PERSISTENT ORGANIC POLLUTANTS IN STELLAR SEA LION

(*EUMETOPIAS JUBATUS*)

A THESIS SUBMITTED TO THE GRADUATE DIVISION OF THE UNIVERSITY OF HAWAI'I IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE OF

MASTER OF SCIENCE

IN

MOLECULAR BIOSCIENCES AND BIOENGINEERING

MAY 2007

By

Kathrin Hülck

Thesis Committee:
Qing X. Li, Chairperson
Shannon Atkinson
Harry Ako
James Carpenter
Eric H. De Carlo

We certify that we have read this thesis and that, in our opinion, it is satisfactory in scope and quality as a thesis for the degree of Master of Science in Molecular Biosciences and Bioengineering.

THESIS COMMITTEE

Chairperson

ACKNOWLEDGEMENTS

I would like to thank my supervisor Dr. Q.X. Li for the opportunity to work in his laboratory, as well as providing funding and support over the last 4 years. I would like to thank my master thesis committee members Dr. S. Atkinson, Dr. H. Ako, Dr. J. Carpenter and Dr. E. De Carlo for their strong support and advice while accomplishing my thesis goals.

Altogether, I would like to thank my laboratory colleges, students and co-workers, especially Chana Hennessee, Dongli Wang, Judith Denery, Sonia Campbell, Renee Harada, Beth Irikura, Karl Yanagihara, and all other CTAHR students, staff and faculty, as well as the MBBE office staff, involved.

I would like to express many thank to my family and friends back home in Germany for their ideas, suggestions, support, and kind words. I could not have done this project without them.

At the Alaska SeaLife Center (Seward, Alaska), I would like to thank Anne Hoover-Miller, Matthew Myers, Carol Stephens and Angie Steeves for their support and teamwork during my visits.

All Steller sea lion tissue samples were obtained through the Tribal Ecosystem Office of St. Paul and Dr. Alan Springer, University of Alaska Fairbanks.

The funding for this research was provided by the Alaska SeaLife Center (Seward, Alaska): Steller Sea Lion Research Program.

PREAMBLE

This master thesis consists of two separate studies on persistent organic pollutants (POPs) in Steller sea lion (*Eumetopias jubatus*). All Steller sea lion tissue samples, such as blubber, liver and kidney, described and discussed in the second chapter of this master thesis were prepared, chemically analyzed, analyzed using GC-ECD, computated and calculated by Dr. Su-Myeong Hong (former address: National Institute of Agricultural Science and Technology, Rural Development Administration, Suwon 441-707, Korea) between approx. 2002 to 2004. Dr. Hong provided the final calculated polychlorinated biphenyl (PCB) concentration data and a first draft of a manuscript. The lead on this study was taken over by me, Kathrin Huelck in 2005. This resulted in a completely new statistical analysis of the received PCB tissue data. From this data new graphs and tables, an altogether newly written introduction, as well as a results and discussion part (excluding original data) were written. The Steller sea lion tissue samples for chapter 2 were collected, analyzed chemically, analyzed instrumentally and computationally as stated in (grammar and sentence structure might have been altered where appropriate):

Chapter 2. Polychlorinated biphenyls in Steller sea lions (*Eumetopias jubatus*) tissues collected from locations in Prince William Sound and the Bering Sea, Alaska

2.3. Materials and methods

2.3.1 Sample Collection

2.3.2 Sample Extraction and Cleanup

2.3.3. Gas Chromatographic Analysis

2.3.4. Identification of PCBs

Additional information on sample collection, preparation and analysis, such as further QA/QC and other quality assuring procedures were not available. All resulting discussions and conclusions for chapter 2, as well as additional discussion and conclusions stated in chapter 4 were based on the original calculations (data set) provided by Dr. Hong.

TABLE OF CONTENTS

LIST OF TABLES

Table

Page

LIST OF FIGURES

Figure

Page

ABBREVIATIONS

ATSDR	- Agency for Toxic Substances and Disease Registry
AMAP	- Arctic Monitoring and Assessment Programme
BHC	- Benzene Hexachloride
BS	- Bering Sea
DDT	- Dichloro-diphenyl-trichloroethane
DDD	- 1,1-dichloro-2,2-bis(p-chlorophenyl)ethane
DDE	- 1,1-dichloro-2,2-bis(p-dichloro-diphenyl) ethylene
HCB	- Hexachlorobenzene
HCH	- Hexachlorocyclohexane
EPA	- Environmental Protection Agency
ESA	- Endangered Species Act of 1973, as amended
FWS	- US Fish and Wildlife Service
HELCOM	- Helsinki Commission (Baltic Marine Environment Protection Commission)
LRT	- Long-range Transport
MMPA	- Marine Mammal Protection Act
NMFS	- NOAA's National Marine Fisheries Service
NOAA	- National Oceanic and Atmospheric Administration
OCP	- Organochlorine Pesticides
PBDE	- Polybrominated diphenyl ether
PCB	- Polychlorinated Biphenyl

PCN	- Chlorinated naphthalenes
PFOS	- Perfluorooctane sulfonate
POP	- Persistent organic pollutants
SSL	- Steller sea lion
SSLRT	- Steller Sea Lion Recovery Team
TBBPA	- Tetrabromobisphenol-A
USFWS	- U.S. Fish and Wildlife Service
WHO	- World Health Organization
UNEP	- United Nations Environment Programme
USDOI	- United States Department of Interior

Chapter 1

Introduction

1.1 Persistent organic pollutants

1.1.1 Introduction to persistent organic pollutants

Environmental pollution is of increasing concern over its effects on environmental and human health. Persistent organic pollutants (POPs) are a diverse group of chemicals that persist in the environment, bioaccumulate through the food web, and pose a risk of causing adverse effects in humans and biota. An adverse effect is any change in body function or structure of cells that can lead to disease or health problems. POPs can be detected in any natural environment around the world (Varanasi *et al.*, 1992; Manz *et al.*, 2001; Strachan *et. al.*, 2001; Zhang *et al.*, 2002). None of these chemical compounds occurs as a natural substance. They were or are produced as end-products or generated as by-products in manufacturing of other chemicals or petroleum products. The use of many POPs (including agricultural formulations or industrial mixtures) started between the 1920s and 1940s.

In 1962, interest was drawn to some of these man-made chemicals by Rachel Carson's book entitled 'Silent Spring'. In her writing she combined laboratory and field data from that time, suggesting that high levels of DDE (1,1-dichloro-2,2-bis(p-dichloro-diphenyl) ethylene), a degradation product of DDT (dichloro-diphenyl-trichloroethane), caused eggshell thinning. DDE and the parent compound DDT were then correlated to unsuccessful hatching of chicks for certain birds of prey species. During the 1970s and 1980s some POPs were either banned or restricted in production and use.

The following thesis includes two studies dealing with two major groups of persistent organic pollutants that were among the banned, polychlorinated biphenyls (PCBs) and organochlorine pesticides (OCPs). All of these compounds have been used extensively either in agricultural or industrial applications (ATSDR, 1998, 2000, 2002, 2005). The chemical characteristics of POPs that once made them valuable and useful in their technical applications are the same ones responsible for their persistence in the environment. Some POPs are very stable and irrepealably enter the environment. POPs have been found in wildlife and humans in pristine areas such as northern Alaska, far removed from point sources of the pollutants. The mechanisms with which POPs enter and distribute in the environment are very complex. Geographical distribution of POPs depends on the initial emission/release (hemisphere), their physical and chemical properties (e.g. persistence and environmental distribution) and abiotic factors such as temperature, pH, light intensity, and surface structure. Local point-sources (or hot spots) are an entry point of initial emissions of POPs into the environment. These emissions usually originate from direct applications of POPs, such as in agricultural pesticide applications, plasticizers in paints or varnishes, which volatilize during wear or after disposal in landfills, leakage from open systems such as transformators or capacitor, and incineration of POPs containing materials. Because industrial applications are concentrated primarily in the Northern Hemisphere the highest POPs concentrations can also be found there (Connell *et al.*, 1999).

Once POPs reach the natural environment (atmosphere, soil, sediment, water and biota) they are distributed by atmospheric and ocean currents. Areas with present or former industrial installations or heavy agriculture often have higher initial POPs concentrations and are considered as so-called "hot spots". These hot spots are also known as point-sources as opposed to non-point pollution (sources of POPs by physical distribution processes). After release into the atmosphere, POPs adsorb to particles and are distributed by ocean and atmospheric currents (Muir *et al.*, 1999). They undergo long-range transport (LRT) from the tropic regions to the polar regions, also called global distillation, multi-hopping or the "grasshopper effect" (Wania and Mackay, 1993). Semi volatile compounds undergo a shift in equilibrium partitioning from gas phase and solid phase (surface), induced by latitudinal and vertical geographical temperature changes (Wania and Mackay, 1996, Wania, 1999). Not all semi volatile compounds condense in the cold environments ("cold-condensation"), only if the compound is neither too volatile for "cold-condensation". POPs included in this study undergo global distillation and "cold-condensation" to different extends, but can be found in high concentrations in the most remote regions of the world. Specifically in cold regions like the Arctic and Antarctic, even though initial emissions from point-sources (agriculture and industry) in those global regions are low or non-existent.

Over time, POPs are dispersed, degraded or transformed in the environment. Abiotic degradation of POPs (chemical breakdown) occurs by hydrolysis in soil, sediment or water, chemical dechlorination, reductive dehalogenation (Matheson and Tratnyek, 1994), and photolysis (WHO 1976). A measure of persistence of a chemical in the environment was established as 'half-life', which represents the calculated time of loss of the first 50% of the chemical.

There are two partitioning coefficients that describe POP distribution in soil and biota, K_{oc} and K_{ow}. (Crosby, 1998). The organic-carbon-distribution-coefficient (K_{oc})

provides an estimate of POPs adsorption onto soil or sediment surfaces. For most organic compounds, the K_{oc} values are usually very large and are therefore generally expressed as their $\log_{10}$ values. The POPs described in this thesis usually show K_{oc} values of Log K_{oc} 3-8, with values > 1 representing physically powerful adsorption forces. The other important coefficient for this study is the n-octanol-water-partition-coefficient (K_{ow}), which gives a measure of the partitioning of a given chemical between water and octanol and is considered representative of partitioning between water and biotic lipids (Veith *et al.*, 1979; Crosby, 1998). Similar to K_{oc} values, the K_{ow} values are also expressed as their $\log_{10}$. For both partitioning coefficients, K_{oc} and K_{ow}, the tendency towards adsorption to solid particles or partitioning into biotic lipids increases with increasing coefficient values.

POPs enter the food chain via two main types of partitioning mechanisms. 'Equilibrium partitioning' stands for the uptake or release of POPs from organisms of lower trophic levels due to osmotic/diffusion processes. For example, POPs enter benthic organisms (known as bioaccumulation) by direct intake and digestion of contaminated sediments or by direct absorption through gills or skin. Another mechanism is "biomagnification", which occures mainly via bioaccumulation of POPs along different trophic levels of food chains (Hornshaw *et al.*, 1983; Rubinstein *et al.*, 1984; Rasmussen *et al.*, 1990; Barron 1995; Harding *et al.*, 1997; Borgå *et al.*, 2001; Muir *et al.*, 2003). At high trophic levels (e.g., fish to marine mammal) 'biomagnification' of POPs along the food chain usually overshadows chemical 'equilibrium partitioning' between the surrounding element and animal.

Biotic degradation and transformations have been shown for many groups of

POPs, but other degradation pathways include aerobic degradation (oxidation) or anaerobic dechlorination (Brown *et al.*, 1987b; Abramowicz, 1990). Bioremediation is a well known concept that causes/enhances POP degradation by microorganism, such as microbacteria (Miles *et al.*, 1971; Williams, 1998; Okeke *et al.*, 2002). Not all POPs are fully transformed or metabolized and some exhibit half-lives in the environment that can range from month to decades.

POPs bioaccumulate in the lipids of all animals, with concentrations and equilibria in the organs differing according to their lipid content. POPs cause adverse (toxic) effects in biota, the lethal dose (50) (LD_{50}) concentrations have been established in laboratory studies for many POPs and other substances (ATSDR 1998, 2000, 2002, 2005). LD_{50} defines the dose of a chemical determined to cause death in 50% of a defined experimental animal population (e.g., mice or rats). Direct correlations have been observed between body fat content and contaminant concentrations for many animal species (Holden, 1973; Kawai and Fukushima, 1988). As described before POPs have been transported to the Arctic by LRT and oceanic vectors. POPs contamination is especially problematic for marine mammals of the arctic food chain, as they are top predators, often occupying high tropic levels and exhibiting high ratios of body lipids (Bruhn *et al.*, 1995; Jenssen *et al.*, 1996; Ross, 2004) as opposed to terrestrial mammals. The large amount of body fat in marine mammals originates in their physiology and development for survival in marine environments, e.g. buoyancy, streamlining and insulation, making them most vulnerable to bioaccumulation of POPs. A variety in concentration levels of POPs in adipose tissues (blubber) of various marine mammals have been reported (Drescher *et al.* 1977; Reijnders, 1980; Boon *et al.*, 1987; Becker *et* *al.*, 2000; Helm *et al.*, 2002; Hall *et al*, 2003; Hobbs *et al.*, 2003; Wang *et al.*, 2007). Adverse effects caused by POPs in marine mammals and other animals include reproduction failure, weakened immune systems, hormone disruption or chronic health disorders (Reijnders, 1986; Law *et al.*, 1989; Olsson *et al.*, 1994, Ross, 1996).

POPs are partially biotransformed in marine mammals, predominantly in liver tissues. The first step in the biotransformation occurs via the cytochrom P450 (CY P450) dependent mixed function oxidase enzymes. Induction of CY P450 is often connected to high body loads (Bruhn *et al.,* 1995; Boon *et al.,* 1997; Troisi *et al.,* 1997; Wolkers *et al.,* 1999; White *et al.,* 2000; Miller *et al.,* 2005) of POPs. Enzyme induction and correlation to POP concentration levels were confirmed in liver biopsies of harbor seal skin (Miller *et al.*, 2005), liver of Steller sea lion and ribbon seals (Teramitsu *et al.*, 2000), liver and blubber of grey seals, harbor porpoise and common dolphins (Boon *et al.*, 1997). Hepatic CY P450, P420 and mixed function oxidases have been found in harbor seals and harp seals (Boon *et al.*, 1987; Troisi and Mason, 1997; Wolkers *et al.*, 1999).

1.1.2 Polychlorinated biphenyls

PCBs, one of the most thoroughly studied group of man-made chemicals, were widely used as plastic additives, cooling fluids, lubricants and pesticide stabilizers. PCBs consist of 209 individual congeners, which differ only by the number and position of chlorine atoms at the basic biphenyl structure (Ballschmitter and Zell, 1980) (Table 1.1). Between 1929 and 1977, approximately 700,000 tons of PCBs (technical mixtures) were produced in the United States. About 99% of PCBs in the US were produced by Monsanto Chemical Company (Sauget, IL) (IARC 1979). Their technical mixtures were named Aroclors. According to their end-use purposes different Aroclor mixtures have different weight % of chlorinated congeners. A summary of 9 different Aroclor mixture end uses is listed in Table 1.2. Reported LD_{50} values in rats were 4250 mg/kg for Aroclor 1242 (Bruckner *et al.*, 1973), 1010 to 1295 mg/kg for Aroclor 1254 (Garthoff *et al.*, 1981; Linder *et al.*, 1974), and 1315 mg/kg for Aroclor 1260 (Linder *et al.*, 1974), respectively.

Higher chlorinated PCBs usually show lower water solubilities, higher K_{oc}, and K_{ow} values, thus exhibiting a greater tendency for adsorption to soil and sediment particles and bioaccumulation in lipids of animals (Table 1.1). Lower chlorinated PCBs show higher water solubilities and lower K_{oc} and K_{ow} values. Thus, these congeners adsorb to a lesser extent to soil or sediment particles and bioaccumulated in lower concentrations in biota (Table 1.1).

The production and use of PCBs have been restricted worldwide and banned in North America as well as Europe in the 1970s and 1980s, respectively. In the United States PCB production and use were restricted in 1976 by US Congress under the Toxic Substances Control Act (TSCA) and the Resource Conservation and Recovery Act (RCRA). Worldwide production, from 1929 to 1989, excluding the Soviet Union, totaled approximately 1.5 million tons (UNEP 1999).

Even though they are considered as one group of POPs, a multitude of chemical characteristics and modes of toxicity and action exist (Castello *et al.*, 1997; Van den Berg *et al.*, 1998; Geyer *et al.*, 2000). PCB 44, 52, 138, 153, 170, 180, and 187 are the most persistent PCB congeners. PCB 77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169, and 189 are known to be the most toxic PCB congeners. Their structure shows unmistakable similarities to 2,3,7,8-tetrachlorodibenzodioxin (2,3,7,8-TCDD) the most potent man made toxic substance. Because they are similar in structure to 2,3,7,8 TCDD, dioxin-like PCBs are thought to share a similar mode of action as well (Aryl hydrocarbon (Ah) receptor). A Toxic Equivalent (TEQs) scheme was established and set up by Van den Berg *et al.* (1998) to report the toxicity of chemicals weighted by their structural similarity to 2,3,7,8-TCDD. Toxic equivalent factors (TEFs) for each dioxin-like PCB were established for birds, mammals and humans individually. These TEFs are then multiplied with the single dioxin-like PCB concentrations to establish TEQs in tissues such as blubber and liver. TEQs can be reported separately for each of the 14 dioxin-like PCB congeners or as ΣPCB-TEQs according to the mixtures found in animals. TEQs offer a method of reporting toxicity normalized to a known toxicity level (2,3,7,8-TCDD), rather than reporting the total number of grams of a mixture of variously toxic compounds.

PCBs are still found in relatively high concentrations throughout the world. Based on monitoring data, especially the polar regions appear to be an environmental sink for PCBs (Muir *et al.*, 1992). Since their restriction, overall PCB emissions into the environment have dramatically declined. Over the last 30 years these recalcitrant chemicals have been dispersed, transformed and metabolized. Thus, intital point-source mixture profiles of these chemical have been surpass and substituted by a more general "homogenizied" profile that can be found in marine mammals independent of their location (Tanabe *et al.*, 1994).

1.1.3 Dichloro-diphenyl-trichloroethane (DDT) and its degradation products

Organochlorine pesticides (OCPs) include the best known pesticide, DDT (dichloro-diphenyl-trichloroethane), as well as its degradation products/metabolites DDD (1,1-dichloro-2,2-bis(p-chlorophenyl)ethane) and DDE (1,1-dichloro-2,2-bis(p-dichloro-diphenyl) ethylene). Among the family of DDTs are two structurally slightly different groups. One consists of p,p'-DDT, p,p'-DDD and p,p'-DDE with two chlorine atoms in para-para positions on the biphenyl ring structure (Table 1.1). The second contains o,p'-DDT, o,p'-DDD and o,p'-DDE with chlorine atoms in ortho-para positions (Table 1.1). DDT is an organochlorine insecticide used in the United States from 1939 to approximately 1972 (Table 1.3). It was very successful and well known for its effectiveness, persistence, low mammalian toxicity and low costs (Metcalf, 1989). At the peak of its popularity in 1962 about 85000 tons were produced annually (Metcalf, 1995); production then decreased by 1971, shortly before its ban, production in the United States was lowered to 2000 tons annually. The worldwide DDT production has been estimated at 2 million tons (Metcalf, 1995). Its applications ranged from agricultural use (e.g., Colorado potato beetle and the European corn borer) to domestic use (e.g., malaria, typhus, lice and mothproofing). The use of technical grade DDT was banned in the United States in 1972 (WHO, 1979; ATSDR, 2002) and later also in other parts of the world. Today DDT is still used, e.g., in many African countries to control Malaria (WHO, 1995). DDD (usually a degradation product of DDT) was also produced in small amounts for use as an insecticide. DDD was banned in the United States in 1972 along with DDT (ATSDR, 2002).

LRT and condensation of DDT, DDE and DDE occurs in cold regions ('cold-condensation"). This LRT leads to a wide dispersion of DDT, DDD and DDE in different environments, such as air, water, soil and biota. DDT can be degraded by photooxidation in the atmosphere or photolysis in soil and water or undergoes slow biodegradation in soil with reductive dechlorination finally forming DDD and DDE. The atmospheric half-lives of DDT, DDD and DDE in range from 1.5–3 days. The soil half-lives vary with temperature, pH, water content, and particle size and were estimated in forest soils in Maine (USA) at about 20-30 years (Dimond and Owen, 1996).

High concentrations can be found in areas such as the Arctic or Antarctic, even though point source pollution in these regions is small. Decreasing concentrations of DDT, and DDD may be a result from transport and chemical degradation. Today, concentrations of DDE exceed those of DDT due to decades of biotic transformation. Slow overall degradation of DDT and its degradation products, and the ban on DDT use in the 1970s in the United States and most of the world have contributed to a decrease in the levels of these compounds in the environment over the past 30 years (Lake *et al.*, 1995, Berggrena *et al.*, 1999, Muir *et al.*, 1999, Norstrom and Muir, 2000).

Low water solubility and a high K_{oc} value (Table 1.1) result in strong adsorption of DDT, DDD and DDE to soil and sediment particles. DDT and its degradation products show high log K_{ow} values of 6.91, 6.51, and 6.02, respectively, for the *p,p'*-isomers. This high lipophilicity, in combination with high persistence, has lead to bioaccumulation of DDT, DDD and DDE in biota. Like other POPS they are known to biomagnify along terrestrial and aquatic food chains, leading to highest overall concentrations in top predators. DDT, DDE, and DDD accumulate in animal lipids and increase in concentrations with increasing lipid level of the organism.

DDT, DDD and DDE show adverse effects in wildlife and humans, with effect to the nervous system, the hepatic system (liver), and to the endocrine system. LD_{50} values in rats ranged from 113 to 800 mg/kg (Cameron and Burgess, 1945; Gaines, 1969; Ben-Dyke *et al.*, 1970). Accumulation of high DDT concentrations can lead to developmental effects and deformities in reproduction systems of male and female animals. Transformation in mammals (studies with mice, rats and hamsters) indicate that the primary route of excretion of DDD and DDE (as well as un-metabolized DDT) is by urine and feces (Gold and Brunk 1982, 1983, 1984). Elimination of these compounds by metabolism ranks as follows: DDE>DDT>DDD.

1.1.4 Hexachlorocyclohexanes

Hexachlorocyclohexanes (HCHs), which were formally known as benzene hexachloride (BHC), are a group of synthetic chemicals with 8 structurally different isomers. Four of these isomers alpha- (α), beta- (β), gamma- (γ) and delta- (δ) were used commercially (Table 1.1). The γ-HCH isomer was the most used and is commonly known as Lindane, an insecticide used to treat corn, oat, wheat and other grains. It was also used in prescription medications for treatment of lice, mites and scabies. Technical grade HCH (60-70% α-HCH, 5-12% β-HCH, 10-15% γ-HCH, 6-10% δ-HCH and 3-4% ε-HCH) was also used as an insecticide (ATSDR 2005) (Table 1.3), but has been banned from production in the US, since 1976. Even though production has stopped, γ-HCH is still imported and used in the US, with about 90 metric tons imported in 2002 (Hauzenberger *et al.*, 2002). Up until 2001, an estimated 500 metric tons of γ-HCH containing pesticides were imported annually by the United States (Hauzenberger *et al.*, 2002).

Compared to other POPs, HCHs have a variety of seemingly contrasting chemical properties. Unlike the other POPs in this study, HCHs have higher water solubility and are known to contaminate groundwater. Nevertheless, they volatilize and enter the atmosphere where they undergo LRT. The log K_{ow} values (Table 1.1) of the different isomers were calculated at 3.72 (γ-HCH), 3.8 (α-HCH), 3.78 (β-HCH) and 4.14 (δ-HCH). These values also are lower than those of other POPs, but HCH isomers also accumulate to a greater extent in biota, especially in the lipids. Unlike other OCPs, the entry point of HCH isomers into the food chains is mainly adsorption from surrounding waters through skin or gills of animals. Therefore, biomagnification in terrestrial food chains is less common than in aquatic food chains. HCH isomers are biodegraded, and subject to hydrolysis and photolysis in soils and water. Photodegradation in the atmosphere occurs with half-lives of 91, 152, 104, and 154 hours for α-HCH, β-HCH, γ-HCH, and δ-HCH, respectively (Chen *et al.*, 1984). Groundwater half-lives for γ-HCH were >300 days and a hydrolysis half-life of 207 days at pH 7 and 25 °C in distilled water was reported by the EPA (1989d). Compared to PCBs and other OCPs the LTR potential and global distribution of α-HCH and γ-HCH were higher (Beyer *et al.*, 2000, Barber *et al.*, 2005). Even though γ-HCH is still used in the US, the worldwide use and production have declined. Due to their high degradation potential by microorganisms, photolysis, hydrolysis and photo-degradation, combined with the decreasing input into the environment, concentrations of the HCHs are declining in soil, water, atmosphere and biota.

HCHs have been connected to causing adverse health effects in wildlife and humans, targeting the nervous system and reproductive systems in both male and female animals. Increases in liver enzymes and fetotoxicity were also observed in laboratory animals. The LD_{50} value for female rats was reported at 91 mg/kg and for male rats at 88 mg/kg (Gaines, 1960). As other OCPs described in this study HCHs also adsorb to soil and sediment particles upon entering the environment. With log K_{oc} reported as 3.57 for α-, β- and γ-HCH and 3.8 for δ-HCH, they how a somewhat lesser affinity for soil and sediments than other POPs (Table 1.1).

1.1.5 Hexachlorobenzene

Hexachlorobenzene (HCB) in a chlorinated hydrocarbon with a variety of applications and properties. It was used in agriculture as a fungicide for treatment of wheat and other grains. In industrial applications it was used for synthetic rubber production, dye production and as a wood preservative (IARC 1979). HCB was restricted for commercial use in the late 1970s, but some small quantities are still produced for laboratory use. Since 1984, HCB has not been sold in the United States for agricultural application and today is released almost exclusively by industrial activities. HCB is a byproduct in the manufacture of organic solvents and pesticides such as terachloroethylene, trichloroethylene and pentachlorophenol (PCP). HCB is considered one of the most persistent man-made chemicals, due to its chemical composition and stability. Its half-life in the atmosphere is estimated at 0.6 years in tropical to subtropical regions, 2 years in temperate regions and about 6.3 years in polar regions. LRT through atmospheric and oceanic currents is likely as is "cold-condensation" from the temperate to arctic regions. The log K_{oc} was calculated at 6.08 (EPA 1981a), a high log K_{oc} values which is consistent with log K_{oc} values of the other POPs listed in this study (Table 1.1). HCB has a strong tendency to adsorb strongly to sediment and soil particles upon entering the atmosphere or oceans. In general, HCB concentrations build up in sediments of fresh-and saltwater systems, such as lakes or oceans. This is supported by a low water solubility of only 5.815 μg/L. The log K_{ow} of 5.73 was reported for HCB (Hansch *et al.*, 1995). The high log K_{ow} suggests high bioaccumulation potential in lipids of biota and biomagnification along the different trophic levels of aquatic food chains (Table 1.1). The Toxic release inventory (TRI) estimated that 6,270 kg of HCB were released into the environment in 1998 (EPA 1995c). Bailey (2001) estimated a total annual emission of HCB into soil at 2,785 kg in the United States. Adverse effects caused by HCB in wildlife and humans are numerous and include neurological, endocrine, systemic, developmental and immunological toxicity. A LD_{50} dose of 3500 mg/kg was reported for rats and 4000 mg/kg for mice (Savitskii, 1964 and 1965).

1.1.6 Heptachlor

Heptachlor is a polychlorinated cyclodiene insecticide and was used in agricultural applications in the US from 1953 to 1974. Today it is not commonly used, but permits still exist and are valid for control of termites and fire ants. Heptachlor is a constituent of technical-grade chlordane, approximately 10% by weight (ATSDR, 2005). Imports of heptachlor into the United States from 1996 to 2003 have declined from 178 tons/year to 70 tons/year (including other chlorinated pesticides). No heptachlor import into the United States was reported for 2004. Heptachlor has a calculated log K_{ow} (Table 1.1) of 6.10 and a log K_{oc} was estimated at 4.34 (Chapman, 1989). In the environment, heptachlor is degraded to heptachlor epoxide, which is more persistent than its parent compound (ATSDR, 2005). Heptachlor, like PCBs, is distributed by atmospheric and oceanic currents, thus undergoes LRT. Heptachlor undergoes photolysis and photosensitized reactions in the atmosphere. In water it undergoes hydrolysis to 1-hydroxychlordene and heptachlor epoxide with an average half-life of 3.5 days (ATSDR 2005). In soil and sediments, heptachlor is degraded at a rate of less than 1% per week (Miles *et al.*, 1971). In laboratory studies, heptachlor showed adverse effects to neurological and developmental schemes, the liver as well as reproduction. The lowest-observed-adverse-effect level (LOAEL) occurred at 0.03 mg/kg/day and LD_{50} at 230 mg/kg/day (Berman *et al.*, 1995). Significance of increasing heptachlor concentrations along aquatic food chains (biomagnification) has been shown by Connell *et al.* (2002), Geyer *et al.* (1982) and Hawker and Connell (1986). This biomagnification is similar to that of PCBs, in that total body burden of Heptachlor increase with increasing trophic levels of aquatic food chains. (ATSDR, 2005)

1.1.7 Production, use and restrictions of persistent organic pollutants in the Russian Federation, China and India

POP production, use and import has been restricted in the United States since the 1970s. Likewise, in Europe use and production have been restricted amongst others by the Helsinki Commission for the Protection of the Baltic Sea (HELCOM). The Helsinki Convention (1974) "on the protection of the marine environment of the Baltic sea area" was signed by the then seven Baltic coastal states and later the "Stockholm convention on persistent organic pollutants" was signed by 182 countries between 2001 and 2002.

PCBs in technical mixtures such as Aroclors were first produced by the Monsanto Co. in the USA in 1929, but were first synthesized in the former USSR (Figure 1.1) in 1934 and produced on an industrial scale in 1939. In the years between 1939 and 1993 the total production of PCBs was estimated to be around 180,000 tons (compared to 700,000 tons produced in the US production 1929 to 1977). Three major PCB mixtures were produced, Sovol (a mixture of tetra- and pentachlorinated PCBs mainly used as a plasticizer in paints), Sovtol-10 (a mixture 9:1 Sovol with 1,2,4 trichlorobenzene, used in transformers) and Trichlorobiphenyl (TCB; mixed isomers of trichlorobiphenyl which were used in capacitors) (AMAP 2003). PCB production was terminated in the Russian Federation between 1987 and 1993, respectively. The total amount of PCBs remaining in PCB-containing equipment was estimated at approximately 27,000-35,000 tons (AMAP 2003).

Technical HCH was banned from use in North America in the 1970s, but was used in China until the 1980s and in India and the former Soviet Union until the 1990s (Li *et al.*, 1999). Zhang *et al.*, (2002) have reported production of approximately 4.9 million tons of HCHs and 400000 tons of DDT in China between the 1950s and 1980s. In India annual use of about 85,000 metric tons of OCPs (DDTs, HCHs and Malathion, ~70%) was reported by Gupta, (2004).

POPs undergo LRT and deposition in Arctic and Antarctic regions through 'cold-condensation" thus rendering their continuous use in many countries of the world questionable (Minh *et al.*, 2006). It is well known that applications in one area can lead to contamination and accumulation of high POPs concentrations in remote locations with no or comparatively low initial POPs emissions.

1.1.8 Analytical Methods and Matrices

In the following section, some general methods used to prepare animal tissues for analysis of POPs will be listed. Different approaches on detection of POPs, e.g., gas chromatography (GC) or high performance liquid chromatography (HPLC) will be further explained.

Soxhlet-, ultrasonic-, supercritical fluid, and accelerated solvent extractions of tissues samples have been used with solvents such as hexane, diethyl ether, acetone, ethanol, acetonitrile or carbon-dioxide. The clean-up of extracts can be accomplished by florisil, neutral alumina or silica gel columns. Analytical detection methods used in previous studies include (among others) GC with electron capture detector (ECD), high resolution GC (HRGC) and ECD, high resolution mass spectrometry (HRMS), GC coupled with mass spectrometry (MS), GC with flame ionization detector (FID). Figures of merit such as the recoveries of the target compounds, standard deviations (STD), sample detection limits (SDL) or method detection limits (MDL) are usually calculated for quality assurance and quality control (Tashiro and Matsumura, 1978; EPA, 1980a, 1980c, 1986f, 1998k, and 1999k; Bristol *et al.*, 1982; William *et al.*, 1984; Burse *et al.*, 1989; Burse *et al.*, 1990; Patterson *et al.*, 1994). A multitude of matrices have been analyzed for POPs in the past, including serum, blood, plasma, human milk, urine, feces, semen, adipose tissue (blubber), skin, hair, liver, kidney, brain, muscle and many more.

Since the 1970s, marine mammals have been included as examination objects in studies on pollution of marine biota by POPs. These studies were continued and extended to pharmacokinetic studies inducing many different species (Aguilar and Borrell, 1991, Severinsen and Skarre, 2000, Wang *et al.*, 2007). Today marine mammals are among the key species used in contaminant studies due to; (1) their longevity and principal position as top-predators in food chains around the world; (2) their adipose tissue (blubber) with overall highest concentrations of contaminants in the body; (3) the reported declines for different species; (4) their position amongst animals hunted for subsistence (as traditional uses of wild foods) thereby passing POPs on to humans and possibly causing adverse heath effects.

1.2 Steller sea lion western stock

1.2.1 Background

1.2.1.1 Habitat

Steller sea lion (*Eumetopias jubatus*) belong to the family of Otariidae with 14 species in 7 genera. Steller sea lions are endemic to several interrelated regions of the Northern Pacific Ocean. The Steller sea lions habitat ranges throughout the Bering Sea (from the west; Russia and Japan to the east; Alaska, Aleutian Islands) and south along the western coast of the North American continent down to north California (Figure 1.1). Two centers of abundance and distribution are the western Gulf of Alaska and eastern and central Aleutian Islands, with about 75% of the historic population (Loughlin *et al.*, 1984). Today about 80% of the western Steller sea lion stock can be found in regions of Kiska-Island and the Kenai Peninsula (Fadely *et al.*, 2006). Steller sea lions are endemic to these areas, but individuals are known to migrate and disperse differently over these regions according to the time of year. They have been divided into an eastern and western stock (Figure 1.1), based on distribution, population dynamics, phenotype and since 1997, on genotype (Bickham *et al.*, 1996; Loughlin, 1997). Males can reach a length of 325 cm (10-11 ft) and can weigh up to 1,100 kg (2,400 lb). The smaller females range around 240-290 cm (7.5-9.5 ft) in length and weight up to 350 kg (770 lb). Females may reach an age of ~30 years and males about 20 years. Steller sea lions inhabit terrestrial as well as marine areas, depending on seasons, foraging behavior and gender. Rookeries (terrestrial mating, pupping and breeding grounds) are occupied by males and females, with females staying longer giving birth to the pups. Non-breeding adult and subadult Steller sea lions (male and females) inhabit haulouts (terrestrial areas used during times other than the breeding season) throughout the year. When in their marine habitat Steller sea lions usually stay near-shore and close to the edges of the continental shelf.

1.2.1.2 Reproduction

Mating and pupping occur on land at rookeries. Females usually give birth to one pup and are known to nurse them approximately 1 year, although weaning times are highly variable. Viable births occur from late May through early July (Pitcher and Calkins, 1981). Females reach sexual maturity at an approximate age of 3 to 6 years and may reproduce up to their early 20s. The male population can be divided into breeding males, non-breeding males (not able to hold territories) and subadults. Males reach sexual maturity between 3 and 7 years of age and physical maturity round the age of 10.

1.2.1.3 Food

Steller sea lions are carnivorous and forage in the marine habitat. Studies since 1975 showed that Walleye pollock is a principal prey in all areas and years (Pitcher, 1981; Sinclair and Zeppelin 2002). Other prey species include squids, herring, flatfish, Pacific cod, Pacific salmon and octopus (Calkins and Goodwin, 1988). Based on data from captive Steller sea lion adults (non-pregnant and non-lactating) individuals need approximately 6-10% of their body weight in food per day (Keyes, 1968). Steller sea lion milk contains about 23-25% fat. The average daily feeding rate for lactating northern fur seals (a close relative to Steller sea lion) was estimated to be 1.6 times higher than for non-lactating females (Perez *et al.*, 1990).

1.2.1.4 Steller Sea Lion Population

Since the 1950s, sporadic population assessments of Steller sea lion had been conducted for various rookeries and haulouts along the Bering Sea, Aleutian Islands (BSAI), and the Gulf of Alaska (GOA). Around the mid 1970s, it was found that a major decline in the Steller sea lion population had taken place. It was first thought the decline was in the eastern population only, because sufficient data and counts for the western population were not available (Braham *et al.,* 1980; Merrick *et al.,* 1987). The eastern population has slowly increased about 3% per year for the last 18 years. Surveys done in 1975-77 and following years (Table 1.4) fast revealed a greater decline (~80%) in the population situated in regions of the Northern Pacific Ocean west of the Gulf of Alaska (USA) (Marine Mammal Commission, 2003). Reasons for the drastic decline of this western population during the 1960s to the late 1980s were thought to include long-term changes in diet, competition for food/fish and fishery related takings.

All though the severe Steller sea lion population decline of the 70s and 80s subsided, between 1991 and 2000, the population still declined at an annual rate of approximately 4% (Loughlin and York, 2001). Due to difficulties of determining accurate population numbers, the Steller Sea Lion Recovery Team (SSLRT) recommended an initial benchmark of 90,000 animals (excluding pups) counted on trend sites in the Kenai-Kiska area (NMFS, 1992). The western stock population of Steller sea lion was estimated to include 22,223 animals in 1996 (Sease *et al.,* 1999). In 1998, their abundance was estimated at 39,031 animals (Ferrero *et al*., 2000). Different surveys of Steller sea lions at the eastern Aleutian Islands in the mid-1970s have shown a major decline in numbers (Table 1.4).

Accurate data and animal counts are difficult for the Steller sea lion population in the different areas of distribution such as Alaska and Russia. Animal counts were either done by air-count and/or ground-count of rookeries and haulouts. Unknown numbers of animals absent from rookeries and haulout were not included and so the numbers shown in Table 1.4 should be taken as an index or minima of population size rather than definite numbers.

1.2.1.5 Natural Mortality

Calkins and Pitcher, (1982) estimated natural mortality rates of Steller sea lions based on data collected in 1975-77. From birth to age three, the estimated rate was 0.53 for females and 0.74 for males. This rate dropped for females to 0.11 by the sixth year and remained at about that level in older age classes. For males the mortality rate decreased to 0.14 in the third year and further to 0.12 in the fifth year of age. Other natural mortality rates were estimated by York, (1994) at 0.22 for ages 0-2, 0.07 at age 3, 0.15 by age 10 and 0.20 by age 20.

1.2.2 Causes for Steller sea lion decline (western stock)

1.2.2.1 Predation

Predation by killer whales during the 1970s, when Steller sea lions were at their highest recorded levels (about 200,000 animals) was probably a minor factor in the species' decline. In studies after the 1970s (1970s-1992), it was suggested that killer whale predation on Steller sea lion could account for 18% of all Steller sea lion mortality (Marine Mammal Commission, 2003). Juvenile Steller sea lions (in contrast to adults) have been shown to be more vulnerable to the risks associated with additional foraging effort (e.g., predation by killer whales). Today, given the nearly 80% decline in the population size of Steller sea lions, it is likely that the impact of similar levels of killer whale predation is more significant and may be affecting the species ability to recover.

1.2.2.2 Prey food shifts

Evidence of major shifts in the abundance of fish and shellfish in the Bering Sea over the past several decades is well documented. Historical data of Steller sea lion stomach contents collected in Alaska may indicate some long-term changes in diet. Walleye pollock was not a major food of animals collected in 1958 (Mathisen *et al.*, 1962), or in 1960 (Fiscus and Baines, 1966), but this shifted towards walleye pollock and Atka mackerel in the 1970s and 1980s (Pitcher et al., 1981; Sinclair and Zeppelin, 2002). It was thought that long-term changes in fish population structure might affect Steller sea lion through nutritional stress. Measurements of Steller sea lion body sizes (girth, weight, and standard length) in the Gulf of Alaska were significantly less for age 1-10 animals sampled in 1985-1986, compared to the 1970s (Calkins and Goodwin, 1988). It is important to notice that prey availability and diversity appear to control the maximum size of the Steller sea lion population (Merrick *et al.*, 1997). Although there is evidence, suggesting that changes in the abundance of major fish species have occurred, the exact causes of these changes and their influence on Steller sea lion population trends are largely unknown. Some authors suggested that the regime shift changed the composition of the fish community and reduced the overall biomass of fish by about 50 percent (Merrick *et al.*, 1995, Piatt and Anderson, 1996). Rosen and Trites (2000) estimated that Steller sea lions would need to consume 56% more pollock than herring for the same net energy intake. Diet studies of captive Steller sea lions indicated that they adjust their intake levels seasonally, with increases in fall and early winter months (Kastelein *et al.*, 1990). These adjustments varied with age and sex of the studied animals. Decreased survival among juvenile Steller sea lions of the western stock (Table 1.4) has also been found (Merrick *et al.*, 191995, Chumbley *et al.*, 1997). Though data are insufficient to isolate nutritional stress of juveniles as the causal factor in the second phase of the decline (1991-2000), it remains a viable hypothesis (ESA 2001). Winship *et al.*, (2002) proposed that larger animals might be able to compensate for changes in prey species and energy requirements. Immature or recently weaned juveniles might be at higher risk to reduced prey biomass, because they have relatively larger metabolic demands and are rather inexperienced in foraging. According to York (1994), the proportion of juveniles lost to the population need not be large (10% to 20%) to result in a population decline. Although food availability is crucial for all animals, it is of utmost importance especially to pregnant and lactating females. At present, it is not possible to measure adult female survival rates accurately to determine to what extent this population may be compromised by prey food shifts. Adult male Steller sea lions might also be affected by a decrease in prey availability, as higher energy demands would lead to longer foraging periods and so compromise their overall health.

1.2.2.3 Subsistence hunting

Under the Marine Mammal Protection Act (MMPA) Alaskan natives are allowed to hunt Steller sea lions for subsistence (Marine Mammal Commission, 2003), even if this species is listed as threatened or endangered. These regulations exist as long as hunting is carried out in a non-wasteful manner and for subsistence purposes only. Steller sea lions remain an important source of traditional food today and systematic fieldwork is required to estimate accurately subsistence harvest. It is further important to determine whether annual-harvest-levels in communities fluctuate significantly from year to year.

1.2.2.4 Fisheries

The Bering Sea region represents one of the biologically and economically most important ecosystems in the United States, providing over 50% of fish and shellfish catches in a multi-billion dollar industry (Fritz *et al.*, 1995, Fritz and Ferrero, 1998). Many Steller sea lions have been taken incidentally by commercial fishing operations. It was concluded that incidental takings of Steller sea lions was a contributing factor to their decline during the 1970s (Perez and Loughlin, 1991). Fisheries could also affect Steller sea lion nutrition not only by causing localized prey depletion, but also by disrupting fish schools during the fishing process. It was estimated that, by 1991, the walleye pollock stock was diminished by 75 percent from their numbers in the 1970s. Commercial fisheries target several of the most important prey species of Steller sea lions. These changes could result in less energy obtained by Steller sea lions due to prolonged foraging for less and smaller prey fish. Changes as observed in the 1970s and 1980s in Steller sea lion growth, reproduction, and survival are all consistent with limited prey availability. About 65% of the prey consumed by marine mammals however is currently not of commercial interest for the Pacific Ocean (Trites *et al.*, 1997). The complexity of ecosystem interactions and limitations of data make it difficult to determine whether fishery removal of prey species has influenced the population of sea lions to the point of species decline (Lowry and Frost, 1985; Harwood and Croxall, 1988).

1.2.2.5 Human activities

Human activities close to Steller sea lion rookeries, haulouts and foraging grounds could have subtle, but significant impacts. Steller sea lions have been shown to continually use areas after harassment, but also temporarily abandoned haulouts after repeated disturbance. The specific impacts on Steller sea lions by various types of disturbance have not been studied specifically, but increased disturbance could have effects on the recovery of the Steller sea lion population. Disturbances originate primarily from cruise ship, ferries, small boats, and aircraft noises. Traffic by federally permitted vessels (within 3 nm of rookeries and haulouts) has been shown to have adverse effects Steller sea lions (ESA, 2000). The exact consequences of such disturbance to the overall Steller sea lion population are difficult to measure.

1.2.2.6 Toxic Substances

Numerous studies have been carried out in the Pacific and Atlantic Ocean on marine pinnipeds such as harbors seals and other seal species in regard to POPs (Drescher *et al.,* 1977; Reijnders, 1980; Miles *et al.,* 1992; Storr-Hansen and Spliid, 1993; Krahn *et al.,* 1997; Becker, 2000; Kajiwara *et al.*, 2001; Aguilar *et al.*, 2002; Hall *et al.*, 2003; Willcox *et al.,* 2004). POPs have been found to induce adverse effects in animals, meaning that they have the capacity to cause harm to living organisms. The adverse effects in marine mammals include immunotoxicity (Ross *et al.*, 1996; Vos *et al.,* 2000), impaired immune functions (Vos *et al.*, 1989 de Swart *et al.*, 1994; Vos *et al.*, 2000; Ross, 2002; Beckman *et al.*, 2003), reproductive failure (DeLong *et al.*, 1973; Reijnders, 1986; Safe, 1990) and other adverse health effects (Bergman *et al.,* 1992; Mortensen *et al.*, 1992; Olsson *et al.*, 1994; de Swart *et al.*, 1994). High concentrations of POPs have been shown to interfere with reproduction in harbor seals from the Dutch Wadden Sea (Reijnders, 1980 and 1986) and are suspected to cause sterility and uterine occlusions in grey seals (Harju *et al.*, 2003). Adult marine mammals are exposed to POPs, which inadvertently leads to exposure of fetuses and pups. Kleivange *et al.* (1995) reported differences in PCB patterns between pups and immature harbor porpoise, which could indicate the presence of a blood/placenta barrier or a selective transport of PCBs. PCBs transfer from mother to pups during the lactation periods, was observed in grey seals and Steller sea lions (Lee *et al.*, 1996; Debier *et al.*, 2003). Thus, POPs and other man-made chemicals are suspected to play important roles in the population decline of marine mammals such as harbor seals in the Netherlands (Mees and Reijnders, 1994). Mőssner and Ballschmitter (1997) proposed the use of marine mammals as global pollution indicators for organochlorine compounds such as PCBs and OCPs. Boyd and Murray (2001) discussed using upper trophic level predators to monitor marine ecosystems.

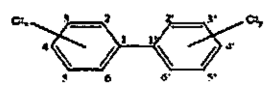
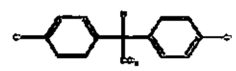
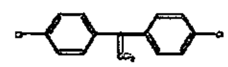
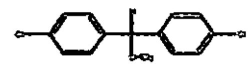
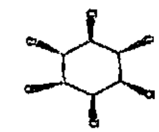
Restrictions in use of POPs has affected their distribution and chemical breakdown in the environment and metabolic elimination in animals and plants have led to overall lower concentrations worldwide (Lake *et al.*, 1995, Berggrena *et al.*, 1999, Muir *et al.*, 1999, Norstrom and Muir, 2000). In the following two chapters, tissues samples of Steller sea lions (such as blubber, liver, kidney and placenta) were analyzed for a variety of POPs including PCBs, o,p- and p,p-DDT, o,p- and p,p-DDD, o,p- and p,p-DDE, HCH isomers (α-, β-, γ- and δ-HCH), heptachlor and HCB. The retrieved data of POPs concentrations were further analyzed statistically and used for spatial and temporal comparisons.

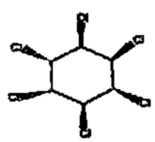
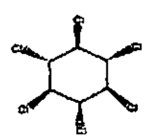
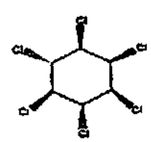
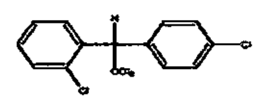
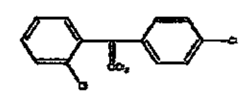
1.3 Goals

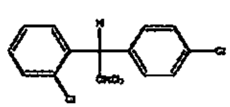
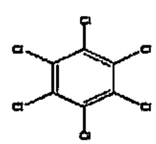
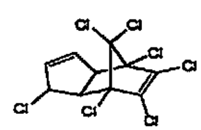
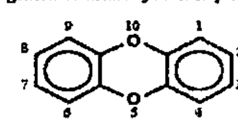
The goal of the research, described in this thesis is to add new data on POPs contamination in Steller sea lions, thus promoting the understanding of current conditions of this species and their habitat. Because they were listed as endangered, the overall goal is to produce data that will aid the recovery of this species.

The long-term goals are to assist the management of the Steller sea lion populations and overall improvement of their habitat. Particularly in the Arctic, marine mammals such as the Steller sea lion are known to be hunted by aboriginal communities to enrich their diet (subsistence hunting and traditional uses of wild foods) (Overton *et al.*, 1990; O'Hara *et al.*, 1999; Bjerregaard and Hansen, 2000; Deutsch and Hansen, 2000; Hansen *et al.*, 2000). Thus, a further long term goal includes aiding evaluations of human health conditions and advancing future management with regard to POPs.

Table 1 .1. Chemical structures, water solubility, log K_{ow}, and log K_{oc} for all OCPs analyzed

Chemical name	Polychlorinated biphenyls	Dichlorodiphenyltri-chloroethane	1,1-dichloro-2,2-bis-(p-dichloro-diphenyl)ethylene	1,1-dichloro-2,2-bis-(p-chlorophenyl)ethane	γ-hexachlorocyclohexane
Common names	PCB	p,p'-DDT	p,p'-DDE	p,p'-DDD	γ-HCH
Number of isomers	209	1	1	1	1
Chemical Formula	$C_{12}H_xCl_y$	$C_{14}H_9Cl_5$	$C_{14}H_8Cl_4$	$C_{14}H_{10}Cl_4$	$C_6H_6Cl_6$
Chemicla Structure					
Water solubility (25°C)	0.00003-0.175 ppm	0.025 mg/l	0.12 mg/l	0.09 mg/l	17 ppm
Log K_{ow}	6.04-8.35	6.91	6.51	6.02	3.72
Log K_{oc}	4.41-7.3	5.18	4.7	5.18	3.57

Chemical name	α-hexachlorocyclohexane	β-hexachlorocyclohexane	δ-hexachlorocyclohexane	1,1,1-trichloro-2-(o-chlorophenyl)-2-(p-chlorophenyl)-ethane	1,1-dichloro-2-(o-chlorophenyl)-2-(p-chlorophenyl)ethylene
Common names	α-HCH	β-HCH	δ-HCH	o,p'-DDT	o,p'-DDE
Number of isomers	1	1	1	1	1
Chemical Formula	$C_6H_6Cl_6$	$C_6H_6Cl_6$	$C_6H_6Cl_6$	$C_{14}H_9Cl_5$	$C_{14}H_8Cl_4$
Chemicla Structure					
Water solubility (25°C)	10 ppm	5 ppm	10 ppm	0.085 mg/l	0.14 mg/l
Log K_{ow}	3.8	3.78	4.14	6.79	6.0
Log K_{oc}	3.57	3.57	3.8	5.35	5.19

Chemical name	1,1-dichloro-2-(o-chlorophenyl)-2-(p-chlorophenyl)ethane	Hexachlorobenzene	Heptachlor	Tetrachlorodibenzodioxins
Common names	o,p'-DDD	HCB	Heptachlor	TCDD
Number of isomers	1			22
Chemical Formula	$C_{14}H_{10}Cl_4$	C_6Cl_6	$C_{10}H_5Cl_7$	$C_{12}H_4Cl_4O_2$ *general structure of dibenzo-p-dioxins*
Chemicla Structure				
Water solubility (25°C)	0.1 mg/l	0.006 mg/l	0.05 mg/l	7.9×10^{-6} - 3.2×10^{-4} mg/l (2,3,7,8 TCDD)
Log K_{ow}	5.87	5.73	6.1	7.2 (2,3,7,8 TCDD)
Log K_{oc}	5.19	6.08	4.34	no data

Log K_{oc} - $\log_{10}$ value of organic carbon distribution coefficient
Log K_{ow} - $\log_{10}$ value of octanol-water partition coefficient

Source: comprised from ATSDR Reports 1998, 2000, 2002 and 2005

Table 1.2. Summary of end use for different Aroclor mixtures

End use	Aroclor								
	1016	1221	1232	1242	1248	1254	1260	1262	1268
Capacitors	•	•				•			
Transformers				•		•	•		
Heat transfer				•					
Hydraulics/lubricants									
Hydraulic fluids			•	•	•	•	•		
Vacuum pumps					•	•			
Gas-transmission turbines		•		•					
Plasticizers:									
Rubbers		•	•	•	•	•			•
Synthetic resins					•	•	•	•	•
Carbonless paper				•					
Miscellaneous:									
Adhesives		•	•	•	•	•			
Wax extenders				•		•			•
Dedusting agents						•	•		
Inks						•			
Cutting oils						•			
Pesticide extenders						•			
Sealants and caulking compounds						•			

IARC 1979

Table 1.3. Compositions of technical grade DDT and technical grade HCH

Technical grade DDT

Compound	% in Mixture
p,p'-DDT	65-73
o,p'-DDT	19-21
p,p-DDD	0.17-4.0
o,o'-DDT	0.1-1.0
o,p'-DDD	0.04

Haller *et al.*, 1945

Technical grade HCH

Compound	% in Mixture
α-HCH	60-70
β-HCH	5-12
γ-HCH	10-15
δ-HCH	6-10
ε-HCH	3-4

Kutz *et al.*, 1991

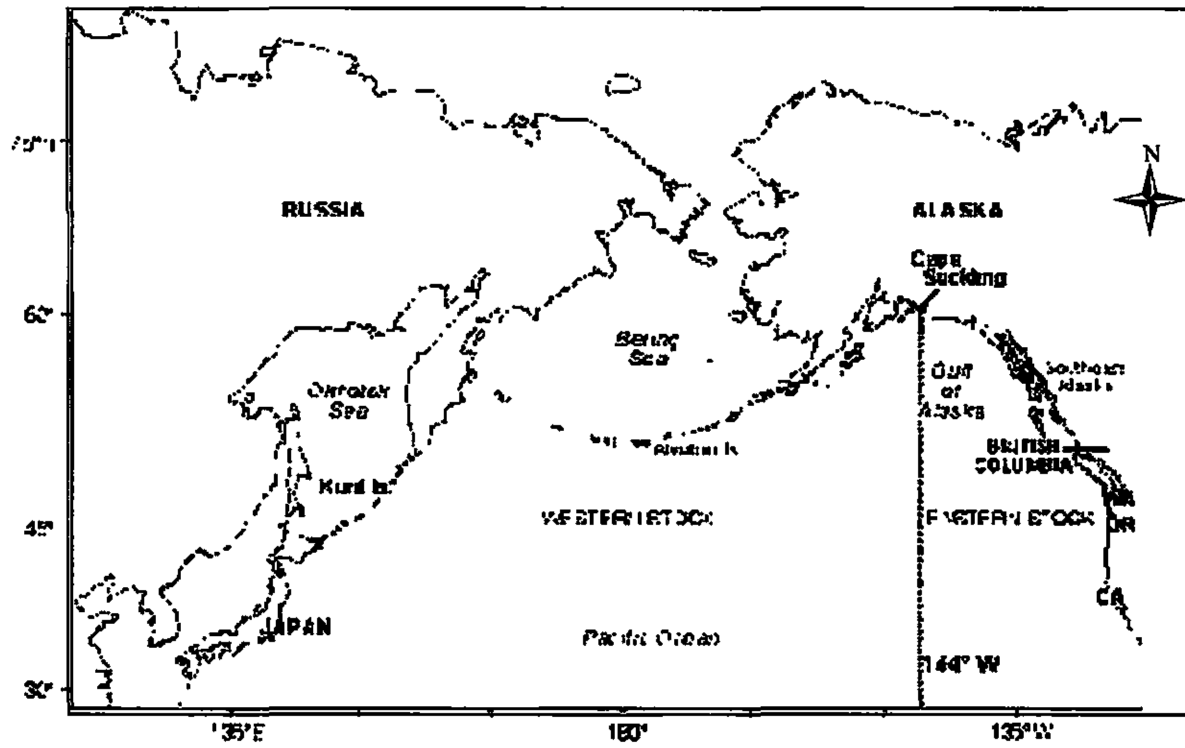
Table 1.4. Counts of adult and juvenile Steller sea lions in the central and western Gulf of Alaska and eastern and central Aleutian Islands (western population)

	Adult/ Juvenile	Adult/ Juvenile	Adult/ Juvenile	Adult/ Juvenile	Adult/ Juvenile
Years	1956-1959	1975-1977	1985	1989	1990
Number	140,115	103,976 (90,000a)	67,617	24,953	27,860

[a] SSLRT: recommended number for time period

Source: Marine Mammal Commission 2003

Figure 1.1. Map of the Steller sea lion population, both eastern and western stock



CHAPTER 2

POLYCHLORINATED BIPHENYLS IN STELLER SEA LION (*EUMETOPIAS JUBATUS*) TISSUES COLLECTED FROM LOCATIONS IN PRINCE WILLIAM SOUND AND THE BERING SEA, ALASKA

Kathrin Hülck, Su-Myeong Hong, Shannon Atkinson, Qing X. Li

2.1. Abstract

The western stock of Steller sea lion (*Eumetopias jubatus*) in the northern Pacific Ocean has shown a decline in numbers of ~80% over the past 30 years and thus was listed under the US Endangered Species Act (ESA) as endangered in 1997. The reasons for the decline are little understood, and pollution by polychlorinated biphenyls (PCBs) is among the possible causes. Concentrations of single PCB congeners have not been reported, thus far. The need for PCB congener specific analyses arises from the fact that these compounds show very distinct levels and mode of toxicity. In this study, 145 individual PCB congeners were determined in tissues of 11 male Steller sea lions from Tatitlek (Prince William Sound) and St. Paul Island (Bering Sea) collected between 1998 and 2001. Besides blubber, liver and kidney samples, placental tissues from 9 female Steller Sea lion were also analyzed. The sum PCB concentrations in Steller sea lion tissue samples were below the immunotoxic threshold of 17 μg/g lipid weight (lw) in the blubber, but two of the animals showed high PCB concentrations in their livers. The values 8.0 μg/g and 6.6 μg/g lw, lie within the physiological toxic threshold of 6.6-11

μg/g lw in the liver of marine mammals. PCBs 90/101, 118, 153 were abundant in all tissue samples. Liver, as the most metabolically active organ, showed a more evenly distributed congener profile than blubber, kidney and placenta. The tissue profiles from the latter had fewer, but higher-chlorinated congeners. Three out of twelve dioxin-like PCB congeners were detected in blubber, liver, kidney and placenta samples. The most potent ones, PCBs 126 and 169 were not detected in any of the tissue samples. The mean toxic equivalents (TEQ) were 9 pg/g lw in kidney, 49 pg/g lw in liver and 22 pg/g lw in the blubber samples. The mean sum TEQ in placentae was 19 pg/g lw. Given the low sum TEQ values, PCBs may induce minor adverse health effects in Steller sea lions examined in the present study.

2.2. Introduction

Polychlorinated biphenyls (PCBs) are a class of synthetic chemicals consisting of 209 theoretical congeners. Each congener shows a different level of chlorination with respect to number and position of chlorine atoms on the biphenyl moiety. Used as transformer dielectric fluids, flame retardants, plasticizers, and pesticide additives large quantities of PCBs were produced and deployed globally since the 1930s (Safe, 1984). All PCB congeners are biologically recalcitrant to a different extent and have been identified as serious threats to public health (Brown *et al.*, 1986; Swain, 1988; Quinn and Allen, 1995) and wildlife (Bruhn *et al.*, 1995; Boon *et al.*, 1997). PCBs were restricted in production and use during the 1970s and 1980s in most industrialized countries. Physical and chemical degradation as well as biological metabolism generate environmental specific congener profiles, which differ greatly from those of the original technical mixtures (Boon *et al.,* 1987; Storr-Hansen and Spliid, 1993; Giesy and Kannan, 1998). In the marine environment, these hydrophobic substances accumulate along the food chain (Rubinstein *et al.*, 1984; Rasmussen *et al.*, 1990; Barron 1995; Kannan *et al.*, 1995; Muir and Savinova, 2003;) and reach highest concentrations in adipose tissues of top predators such as marine mammals (Krahn *et al.*, 1997; Severinsen and Skaare, 2000; Storelli and Marcotrigiano, 2003).

One such animal is the Steller sea lion (*Enumetopias jubatus)*, which is endemic to the northern Pacific Ocean, where it inhabits both the eastern and western regions. The western stock has shown an ~80% decline in numbers over the past 30 years, which led to its listing as "threatened" under the US Endangered Species Act in 1990, and as "endangered" in 1997. The reasons for of this decline are little understood, but chronic exposure to pollutants such as PCBs is among the possible causes. PCBs exhibit a broad range of toxic responses in wildlife that lead to adverse effects, such as to early developmental defects (teratogenesis) (Barron *et al.*, 1995), immune impairment (Swart *et al.*, 1994; Ross *et al.*, 1996) and reproduction failure in various marine mammal species (Helle *et al.*, 1976; Reijnders *et al.*, 1986; Hooper *et al.*, 1990; Vos *et al.*, 2000). Large body-lipid reserves in addition to their top level position in the arctic food chain make Steller sea lions highly susceptible to biomagnification and bioaccumulation of persistent organic pollutants. Despite abundant documentation of PCB pollution and its adverse effects on marine mammals, data on PCBs for Steller sea lions are rare (Varanasi *et al.*, 1992; Lee *et al.*, 1996, Krahn *et al.*, 1997 and 2001) (Table 2.1.). Lee *et al.* (1996) were among the first to show PCBs in Steller sea lion adipose tissue (blubber) and liver. Because the POPs are highly lipophilic (fat loving), their concentrations for manimal tissues are in general normalized to dry weight (*dw*) and lipid weight (*lw*). The dry weight is based on the weight of the dried tissue samples. POPs concentrations in this study are usually calculated based on 1 g of dry tissue sample. The lipid weight (*lw*) refers to the extractable lipids extracted from tissue samples by an chemical extraction processes. Lipid weights are determined by weight and the POPs concentrations converted from the original dry weights. Samples collected between 1976 and 1978 showed total PCB concentrations from 5.7-41 μg/g *lw* in male Steller sea lions. Now almost 30 years later the question of PCB accumulation in the environment and bioaccumulation/ biomagnification of these compounds in biota remains.

Another imminent question is the toxicity and resulting risk of persistent organic pollutants leading to reproductive failure and other major adverse effects in animals and their offspring. In general, but especially for declining species populations, reproduction is an important element of securing and increasing future animal numbers. Because PCBs are known to have toxic effects on adult animals (Reijnders *et al.*, 1986; Olsson *et al.*, 1994; Barron *et al.*, 1995), any maternal exposure also automatically exposes any fetus via blood circulating through the placenta. Fetuses are especially susceptible to POPs due to their lacking defense mechanism during early developmental stages and exposure to POPs can lead to teratogenesis. The most common problems associated with POPs exposure in mother animals are uterine blockages and spontaneous abortions, and low birth weights and overall weakness for pups (Hutchinson and Simmonds, 1994). Delong *et al.*, (1973) reported premature birth in California sea lions connected to organochlorine bioaccumulation. A study by Harju *et al.*, (2003) suggests that PCBs undergo transplacental movement into the fetus. In the present study, nine unique placenta samples from free-ranging female Steller sea lions were available for PCB analysis in addition to adipose (blubber), liver and kidney tissue from male Steller sea lions. Although the metabolic activity in placentae is not as pronounced as in liver or kidney tissues, metabolism of PCBs in Steller sea lion placenta might alter PCBs concentrations and profiles (Pasanen, 1999).

Numerous present-day studies report that high PCB concentrations and the resulting toxic effects are still found in many marine mammals around the world (Bernt *et al.*, 1999; Becker, 2000; Severinsen *et al.*, 2000; Aguilar *et al.*, 2002; *Helm et al.*, 2002; Kucklick *et al.*, 2002; Le Boeuf *et al.*, 2002; Fossi and Marsili, 2003; Shaw *et al.*, 2005; Del Toro *et al.*, 2006). In recent years, increasing attention has been paid to risk assessment and management of animal populations and habitats, with focus on persistent organic pollutants and their adverse effects. A number of toxic equivalency factor (TEF) schemes have been developed, which give a measure of the relative toxicity of different organic compounds with respect to 2,3,7,8-terachlorodibenzo-*p*-dioxin (TCDD), the most potent man-made organic pollutant (Van den Berg *et al.*, 1998; Kannan *et al.*, 2000). These schemes are based on the assumption that there is a common binding mechanism to the aryl hydrocarbon (Ah) receptor, which is similar amongst many persistent organic compounds of similar structures (Van den Berg *et al.*, 1998). TEFs are used to calculate toxic equivalents (TEQs).

$$\text{PCB congener concentration (pg)} \times \text{TEF} = \text{TEQ (pg)} \qquad \text{(Equation 1.1)}$$

Thus, by establishing TEQs (normalizing dioxin-like organic pollutant concentrations to

TCDD) different tissue types can be evaluated, classified, and compared against each other. The complexity of PCB mixtures and their diverse distribution in the environment make risk assessment and evaluation of their presence very difficult. Conclusions may be biased if based only on the total amount of PCBs. Individual PCB congeners should be quantified in order to take their differences in toxicity and mode of action into account (Castello *et al.*, 1997; Van den Berg *et al.*, 1998; Geyer *et al.*, 2000). While all 209 congeners are of interest, attention has been paid especially to dioxin-like PCB congeners 77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169, and 189. Being structurally similar to TCDD, this group of PCBs has its own set of TEFs (Human/mammals, fish and birds) established by Van den Berg *et al.*, 1998. In the present study, tissues from Steller sea lions were examined for the presence, concentrations and toxic potency of PCBs for risk characterization and applications to future eco-toxicological management.

The particular goals of this study were to

1) Determine the state of the western stock of Steller sea lions in regards to PCB pollution (by comparing 145 PCB congener concentrations in adipose tissue (blubber), kidney, and liver of male Steller sea lions from two locations in Alaska),

2) Establish the presence of PCBs in nine placentae samples from free-ranging Steller sea lions collected along the Aleutian Islands,

3) Assess the presence and toxic potential of dioxin-like PCB congeners in each tissue type and location by means of their TEQs.

2.3. Materials and methods

2.3.1. Sample Collection

Male Steller sea lion tissue samples were obtained from subsistence-hunting from May 1998 to May 2002. Samples from two locations, St. Paul Island (Pribilof Islands) in the Bering Sea (BS) and near Tatitlek in Prince William Sound (PWS), were obtained (Figure 2.1.). Adipose tissue (blubber, n=11), liver (n=10) and kidney (n=9) samples were excised, wrapped in aluminum foil, and immediately frozen at –20 °C until chemical analysis (Table 2.2.). Eight fresh placenta samples found on rookeries at several of the Aleutian Islands and one from Hokkaido, Japan were collected and immediately frozen at –20 °C until chemical analysis (Table 2.3.). Only placentae that were fresh and absent of visible or olfactory decomposition were collected.

2.3.2. Sample Extraction and Cleanup

PCBs in the tissues were analyzed in triplicate as previously described (Miao *et al.*, 2000) with the following modifications. Acidic alumina (60-325 mesh, 4 g) was used as a retainer for lipids and placed on a 2-μm frit filter at the outlet of a 20-ml supercritical fluid extraction cell. Freeze-dried Steller sea lion tissues (1.5 g of liver, kidney or blubber; 1.0 g of placenta) were then added. PCB 209 was added (100 μl of 100 pg/μl) to the sample in the extraction cell and completely mixed. PCB 209 was selected as a surrogate because it was non-detectable in the Steller sea lion tissues. Not all extractable lipids could be retained by the acidic alumina in the extraction cells of the blubber samples. The remaining lipids in the blubber extracts were removed using gel permeation chromatography (GPC). The GPC column (30 cm x 29.5 mm i.d.) was packed with 60 g (dry weight) of 200 to 400-mesh Bio-Bead S-X3 beads (Bio-Rad Laboratories, Hercules, CA, USA). The GPC column was loaded with a tissue extract (~2ml) and eluted with 250 ml of methylene chloride-hexane (1:1, v:v). The first 100 ml containing lipids were discarded and the following 150 ml collected for further cleanup. The eluates were reduced under a gentle nitrogen-gas stream and transferred into hexane.

Liver and kidney extracts were treated with concentrated sulfuric acid (5 x 3 ml) to remove lipids, washed with 5% NaCl (5 x 3 ml) and dried using anhydrous Na_2SO_4. Silica gel (40 μm, J.T. Baker Inc.) and neutral alumina (60-345 mesh, Fisher Scientific Inc.) were dried at 350-450 °C for 12 h approximately. Multilayer glass chromatographic columns (45 cm x 10 mm i.d.) were fitted with Teflon stopcocks and filled bottom to top with 3 g of 3% deactivated silica gel, 2 g of 6% deactivated neutral alumina and 1 cm height of anhydrous Na_2SO_4. Sulfuric acid-treated extracts and GPC eluates (1 ml each) were each loaded onto a column and eluted with 20 ml of hexane.

2.3.3. Gas Chromatographic Analysis

After each fraction was concentrated to 200–500 μL in isooctane, PCBs were analyzed with high resolution gas chromatography-electron capture detection (HRGC-ECD) as previously described (Miao *et al.*, 2000) with the following modifications. The two GC capillary columns used were 50 m×0.25 mm (i.d.) coated with 0.25-μm film thickness ZB-1 (Phenomenex, CA, USA), and DB-XLB of the same dimension (J&B, CA, USA) respectively. The oven temperature was programmed from 120 to 270 °C at a rate of 2 °C/min, and a final hold time of 5 min. Injector and detector temperatures were 270 and 320 °C, respectively. The recoveries of PCBs from the samples ranged from 75 to 112% after corrections for background PCB concentrations. Procedural blanks were carried out through the entire analytical procedure to check for interference and contamination. The limits of detection (LOD) were in a range of 1–50 pg/g for PCB congeners dependent upon the degree of chlorination and approximately 5 pg/g for most individual congeners. The recoveries of the surrogate ranged from 72% to 86%. Reported PCB concentrations were not corrected according to the recoveries of the surrogate. Their IUPAC numbers throughout this article represent the different PCB congeners. A system of a CP-8400 autosampler, Varian CP-3800 GC and Saturn 2000 ion trap MS was used for chemical confirmation. The PCB standard mixtures C-CS-01 to C-CS-05 were purchased from AccuStandard, Inc. (New Haven, CT, USA) and contained 145 PCB congeners.

2.3.4. Identification of PCBs

PCB peak identification was achieved by accurate calibration of retention times based on the concept of the relative retention index, *Pi*, and retention times of the selected PCB internal standards. The *Pi* was calculated from the predicted retention times with the database of the retention parameters and the migration equations (Zhang *et al.*, 2002). By comparison of the calibrated and expected retention times of PCBs, the method was suitable for comprehensive, quantitative and congener-specific analyses of PCBs (Zhang *et al.*, 2002). Retention index values referred to in this study were calculated and used for identification. The retention times of 12 PCBs (PCBs 1, 9, 10, 16, 62, 90, 131, 185, 198, 206, 207, and 209) were measured with an accuracy of ±0.01 min. Some PCBs, however, co-eluted with other PCBs under the analytical conditions used. The coeluted PCBs were congeners 4/10, 7/9, 16/32, 29/54, 41/71, 47/48/75, 52/73, 56/60, 66/80, 93/95, 67/100, 81/87/111, 90/101, 115/116/117, 122/131/142, 124/147, 135/144, 138/163/164, 157/201, 162/183, 170/190, 171/202, 195/208, and 196/203.

2.3.5. Statistical Analysis

The total PCB concentrations in the blubber, liver and kidney samples were treaded with JMP 5.1 statistical software (SAS Institute Inc.) in order to compare data between Tatitlek (PWS) and St. Paul Island (BS). Due to the small sample sizes of all tissue types, it was not possible to determine if the data were normally distributed. Thus, the nonparametric median and interquartile range (iq R) of PCB concentrations were given and the Wilcoxon-Mann-Whitney test chosen for the statistical data analysis. The level of significance in the Mann-Whitney U test was set to be $p<0.05$. Box plot graphs were generated with Minitlab®Release14.20 (Minitlab, Inc.).

2.4. Results and Discussion

2.4.1. Total PCB concentrations and congener profiles in blubber, liver and kidney for Tatitlek (PWS) and St. Paul (BS)

Among the 145 PCBs 51 single or co-eluded congeners were detected in the blubber tissue, 66 in liver tissue, and 38 in kidney tissue. The total PCB concentrations in the Steller sea lion tissues were reported on a dry weight (*dw*) and lipid weight (*lw*) basis. Median concentrations of sum PCBs were 0.06 µg/g *dw* in kidney, 0.2 µg/g *dw* liver and 0.8 µg/g *dw* in the blubber samples, respectively (Table 2.4). The median concentrations in lipid weights across the tissues ranged from 1.3 µg/g *lw* in kidney and 2.3 µg/g *lw* in blubber to 3.7 µg/g *lw* in liver tissues (Table 2.1; Table 2.4). Figure 2.2., shows a box-plot of the median PCB concentrations by tissue type as well as concentration ranges (min.- max.) for each location. Overall, the median PCB concentrations for the different tissue samples from the two locations Tatitlek (PWS) and St. Paul (BS) did not differ significantly (p<0.05).

Data from the present study lie about one order of magnitude below (Table 2.1) the concentrations of total PCBs of up to 23 µg/g *lw* in blubber of Steller sea lions from PWS and BS first reported by Varanasi *et al.,* (1992). The median concentrations of sum PCBs in the present study were also lower than the previously reported average in blubber (5-17 µg/g *lw*) but comparable to the liver (4-9 µg/g *lw*) of Steller sea lions from Alaskan waters and the Russian part of the Bering sea reported by Lee *et al.*, (1996). Krahn *et al.*, (1997) published PCB concentrations in Steller sea lion from Southeast Alaska of 6.6 µg/g *lw* (Table 2.1).

Comparison of data from different laboratories and decades is especially difficult. This arises from differences and advances in analytical methods. Differences in locations also have to be taken into account, as well as multiple definitions of 'sum PCBs' with great varieties of the number of congeners. The differences between the publications of Varanasi et al (1992) and Lee *et al.*, (1996) and the present study are more than one order of magnitude and therefore are likely real. Only Krahn *et al.*, (2000) reported similar total PCB concentrations (1.4 µg/g *lw*) in blubber of Steller sea lion from PWS and in blubber of animals from Southeast Alaska (1.6 µg/g *lw*).

Due to the rarity of Steller sea lion tissue samples, little published work is available for comparison. Therefore, PCB concentrations in the literature are often compared across different, but closely related species. As can be seen in Figure 2.3., Steller sea lion are most closely related to Northern fur seals and California sea lions. The Northern fur seal habitat in the Pribilof Islands also overlaps with haulouts of Steller sea lion. Comparing the data range from the present study with those of Northern fur seals from the Pribilof Islands (Krahn *et al.*, 1997; Laughlin *et al.*, 2002) indicates that the PCB concentrations are within the same order of magnitude. Two studies of PCB concentrations in California sea lions were chosen for comparison (Table 2.1). The concentrations of the two studies (Kajiwara *et al.*, 2001 and Kannan *et al.*, 2004) are about 1-2 orders of magnitude higher than those in the present study. The habitat of Steller sea lions and the California sea lions overlaps in the northern part of California, but Steller sea lion tissues from the present study originated from areas much further northward (Figure 2.1.). The high concentrations of PCBs in blubber of California sea lion (Kajiwara *et al.*, 2001 b) can be explained by the proximity to the continent of California animals versus animals from Alaska and the Bering Sea. The greater industrialization of California also results in great contamination with POPs. Steller sea lions share the same order (*Carnivora*) but harbor seals, bearded seals and northern elephant seals are in different families (*Otariidae* vs. *Phocidae*) in the animal kingdom. Harbor seals are often used as surrogates or test populations for PCB pollution studies. Our data show great differences in comparison to studies from Krahn *et al.*, (1997) and Wang *et al.*, (2007). Bearded seals (Krahn *et al.*, 1997) and harbor seals (Krahn *et al.*, 1997; Wang *et al.*, 2007) showed total PCB concentrations in blubber and liver of about one order of magnitude below the data of Steller sea lion found in similar regions of the present study. Even though harbor seals are one of the most studied marine mammals in these geographical regions and can act as a surrogate for other marine mammals, their data have to be taken cautiously when used for comparison with species from the family *Otariidae*, as they appear to be a conservative surrogate for contamination studies.

Lee *et al.* (1996) reported that average concentrations of PCBs and DDTs in the liver of male Steller sea lions from the Russian Bering Sea, were significantly lower than those from Alaska. They reported PCB levels from 4 µg/g *lw* in liver of female and 9 µg/g *lw* in liver of male Steller sea lions collected from Alaska, compared to 4.0 µg/g *lw* in liver collected from female Steller sea lion in the Russian Bering Sea (Lee *et al.*, 1996). Concentrations of specific PCB congeners were not reported. In the present study, the median PCB concentrations in the kidney, liver, and blubber samples collected from Tatitlek, were 0.5, 3.2 and 2.4 µg/g *lw* respectively. The median PCB levels in the kidney, liver, and blubber samples, collected from St. Paul Island, Bering Sea, were 1.3, 4.4 and 1.6 µg/g *lw*, respectively, (Figure 2.4.). Taking into account the concentration range for each location by tissue type, we found no significant differences in PCB concentrations between Steller sea lion tissues from St. Paul Island and Tatitlek (Figures 2 and 4).

Pentachlorobiphenyls and hexachlorobiphenyls were the prevalent PCB homologues in kidney blubber and liver samples (Figure 2.5.). The data presented in Table 2.5, show average percentages of PCB homologues, therefore it is difficult to say which of these two homologue groups outweighs the other. Figure 2.5., shows the PCB homologues in three different Arochlors 1260, 1254 and 1242 (Monsanto Chemical Co., IL). These three Arochlor mixtures were chosen because they were the most widely used among the nine Arochlors produced by Monsanto in the US (IARC 1979). Weathered and bioaccumulated patterns of PCBs in each of the Steller sea lion tissues differed greatly from those in the technical mixtures 1260 and 1242 (Figure 2.5.). Only Aroclor 1254 shows pentachloro- and hexachlorobiphenyls as their mayor homology groups. The Arcolor 1254 is somewhat similar to those of the Steller sea lion blubber, liver and kidney, but show a much higher percentage of pentachlorobiphenyls than the tissue sample.

Prominent PCB congeners in the blubber tissues were 90/101, 118, 138/163/164, 153, and 154, which accounted for 60% of the total PCBs detected (Figure 2.6.). Dominant congeners in the kidney samples were PCBs 90/101, 115/116/117, 153, 154 and 180, which accounted for 45% of the total PCBs detected (Figure 2.6.). For the liver tissues, predominant PCBs were 19, 41/71, 92, 146, and 149 which accounted for 25% of the total PCBs detected in the tissues. Overall the number of congeners detected in the liver samples was higher than in the other two tissue types (Figure 2.6.). The five dominant PCB congeners in the liver samples accounted for only 25% of the sum PCBs, whereas the dominant five PCB congeners in blubber and kidney accounted for 60% and 45%. In comparison to the blubber and kidney, the concentrations of the individual congeners in the liver samples were more evenly distributed across the tri- to octa-chloroPCBs. The blubber and kidney samples shared dominant PCBs 90 /101, 153 and 154, which, however, were not the dominant PCB, congeners in the liver. This difference may be related to the more active liver metabolism, which renders this organ a primary recipient of pollutants (Becker 2000). The kidney is also known to be metabolically active, but to a lesser extent than the liver which might explain the similarities to the blubber tissues with respect to PCB distributions.

Le Boeuf *et al.* (2002) reported that PCB concentration levels in blubber of California sea lions were above the immunotoxic threshold of 17 µg/g *lw* (Kannan *et al.*, 2000). This was consistent with the findings from Reijnders, (1986) and De Swart *et al.*, (1995) which also found PCB concentration levels above the immunotoxic threshold during feeding studies with harbor seals from the North Sea. The total PCB concentrations in the kidney, liver and blubber samples measured in the recent study were below the immunotoxic threshold (Kannan *et al.*, 2000). The median concentration in the liver (3.7 µg/g *lw*) of Steller sea lions was below the physiological toxic threshold for marine mammals of 6.6-11 µg/g *lw*. Interestingly, in two of the liver samples from animals SSL-2 (5) and SSL9802SNP the PCB concentrations (8.0 µg/g and 6.6 µg/g lw) were within the physiological toxic threshold set by Kannan *et al.*, (2000). These two male animals likely experienced adverse effects of immune and reproductive functions in connection with high PCB concentrations.

2.4.2. Total PCB concentrations and profiles in placenta samples

PCBs were analyzed in eight Steller sea lion placenta samples collected along the Aleutian Islands and one from Japan (Figure 2.1.; Table 2.3.). From the selected 145 PCB congeners, 35 were detected in the placenta tissues (Figure 2.6.). No PCB data were previously reported in placentae from Steller sea lions. The median PCB concentration in the placenta tissues was 1.2 µg/g *lw* with a range of 0.4 to 6.6 µg/g *lw* (Table 2.1). The dominant homologues in the placenta samples were tetra- and penta-chlorobiphenyls

(Figure 2.5.). POPs in marine mammals tend to accumulate during the juvenile years, both in males and females, because the intake of contaminants usually exceeds metabolism and excretion. The concentrations of PCBs in females often decrease when large quantities of PCBs are transferred to their pups during pregnancy (Tanabe *et al.*, 1985; Aguilar and Borrell, 1988; Storelli and Marcotrigiano, 2000b) and, to a greater extent, during lactation (Nakata *et al.*, 1995; Lee *et al.*, 1996). The placenta samples in this study showed the overall lowest concentrations of PCBs compared to blubber, liver and kidney samples from Tatitlek or ST. Paul Island. Martineau *et al.*, (1987) showed increased concentrations of POPs in adult beluga whales as opposed to in juvenile animals. Aguilar and Borrell, (1988) reported similar findings in fin whales, with contaminant levels increasing with age but reaching a plateau in older individuals. Because similar accumulation patterns are likely to exist in adult Steller sea lion blubber, liver and kidney, data from this current study should only be cautiously compared to those from female Steller sea lion.

PCBs 90/101, 118, 153, 154 and 199 (45% of sum PCBs) were the most abundant congeners in the placenta samples (Figure 2.6). Thus, the placentae shared four of the five dominant congeners with the blubber tissue. This, however, is a general profile established as an average of all the nine placentae. The congener profiles varied widely among the nine placenta samples (Figure 2.7). Even though five of the nine samples were collected in close locations at Seal Rock, they do not appear to share the same congener profiles. Unfortunately, no information on age, season and number of pups was available for the female placenta tissues. It is difficult to define the reasons for these very different PCB congener profiles (Figure 2.7). Because PCB congeners were found in the placenta, they were likely transferred to the fetus. It would be most interesting to examine PCB congener profiles in placenta, and tissues of fetuses and corresponding mother Steller sea lions to define the role of the placenta as a possible barrier for PCB transfer, as well as its selectivity of PCB congeners.

2.4.3. 2,3,7,8-TCDD toxic equivalents (TEQs)

Individual PCB congeners differ greatly in their toxic potency (Castello *et al.*, 1997; Van den Berg *et al.*, 1998; Geyer *et al.*, 2000). Non- and mono-ortho chloro-PCBs (dioxin-like) exhibit similar properties to 2,3,7,8-TCDD, whereas the non-ortho PCB congeners are more toxic than mono-ortho ones. However, little information on TEQ values in the tissues of Steller sea lions was found in the literature. Mammal-specific TEFs developed by Van den Berg *et al.*, (1998) were used to estimate the TEQs of non- and mono-ortho PCBs in the issues of Steller sea lions. Only four (PCBs 81, 105, 118, and 157) out of the 12 dioxin-like PCBs were detected in the current study (Table 2.5). The mean total TEQs were 9, 49, and 22 pg/g *lw* in the kidney, liver, and blubber samples, respectively (Table 2.5). The TEQ values were more than one order of magnitude below the threshold level of 520 pg/g lw for toxic effects determined by Kannan *et al.*, (2000). This suggests that exposure to PCBs may contribute to only a low toxicity in Steller sea lions. Wang *et al.*, (2007) showed comparably low TEQs for harbor seal liver and blubber tissues (12-22.5 pg/g and 8.6-173 pg/g *lw*) from similar locations. The detected dioxin-like PCBs (81, 105, 118, and 157) in this study are among the lowest TEFs of all dioxin-like congeners (0.0001, 0.0001, 0.0001 and 0.0005). Overall, PCBs 126 and 169, which are thought to have the highest toxic effects (TEFs of 0.1 and 0.01), were not detected in any of the Steller sea lion tissues from the two locations. Even in the male Steller sea lions with high blubber PCB concentrations such as SSL-4 (2.9 μg/g *lw*) and SNPSLS9901 (2.5 μg/g lw), PCBs 126 and 169 were not detected. A TEQ threshold for placenta has not yet been established. A sum TEQ was calculated for placenta samples (19 pg/g lw) based on the TEFs established by Van den Berg *et al.*, (1998). This was found to be above the TEQ of the male Steller sea lion kidney (Table 2.5). Given the low sum TEQ values, PCBs may induce minor adverse effects on Steller sea lions.

2.5. Acknowledgements

All samples were obtained through the Tribal Ecosystem Office of St. Paul and Dr. Alan Springer, University of Alaska Fairbanks, Ms. Angie Steeves assisted with manuscript preparation. This study was supported by Alaska SeaLife Center Steller Sea Lion Research Program with funds from National Marine Fisheries Services.

Table 2.1. Selected literature of marine mammals from the Northern Pacific Ocean and their sum PCB concentrations (µg/g *lw*)

Species	Location	Tissue	Gender/Age (y)	sum PCB µg/g lw	n	Publication
Steller sea lion	Prince William Sound (Tatitlek)/ Bering sea	blubber	M / juvenile-adult	2.3 (1.1-2.9)	9	Present study[a]
	(St. Paul Island)	liver		3.7 (2.6-7.9)	10	
		kidney		1.3 (0.14-2.2)	9	
	Aleutian Islands	placenta	F	1.2 (0.4-6.6)	9	
Steller sea lion	Prince William Sound / Bering sea	blubber	M, F / juvenile-adult	23 ± 37	8	Varanasi et al. 1992
Steller sea lion	Alaska	blubber	M / 0.9-12	17 ± 12	12	Lee et al. 1996
			F / 2-25	5.3 ± 5.4	17	
		liver	M / 2-12	9.0 ± 6.1	13	
			F / 2-25	4.0 ± 4.0	15	
	Bering Sea	liver	M / 2-16	4.0 ± 1.5	14	
Steller sea lion	Southeast Alaska	blubber	M, F / juvenile	6.6 ± 0.5	3	Krahn et al. 1997[b]
Steller sea lion	Prince William Sound	blubber	M, F / juvenile	1.4 ± 0.7	19	Krahn et al. 2001
	Southeast Alaska			1.6 ± 0.6	10	
Northern fur seal	St. Paul (Pribilof Islands)	blubber	Pups	4.3 ± 1.5	5	Loughlin et al. 2002
			M / juvenile	3.0 ± 1.1	10	
	St. George	blubber	Pups	1.6 ± 0.8	8	
			M / juvenile	2.4 ± 0.9	10	
Northern fur seal	Pribilof Islands	blubber	M / juvenile, subadult	1.3 ± 0.5	7	Krahn et al. 1997[c]
Bearded seal	Norton Sound (Bering sea), AK	blubber	M / unknown	0.2 ± 0.10	5	
Harbor seal	Prince William Sound	blubber	M / pup, adult	0.6 ± 0.1	3	
Harbor seal	Prince William Sound	blubber	M / juvenile	0.2 ± 0.1	14	Wang et al. 2006
		liver		0.3 ± 0.3	3	
		blubber	F / juvenile	0.1 ± 0.1	4	
		liver		0.2 ± 0.2	4	
	Kodiak Island	blubber	M / juvenile	0.4 ± 0.05	6	
		liver	F / juvenile	0.2 ± 0.05	7	
California sea lion	Northern California Coast	blubber	M, F / adult, subadult	461 ± 533	13	Kajiwara et al. 2001 [b]
		liver		166 ± 152	9	
California sea lion	California	blubber		23.1 (2.4-410)	36	Kannan et al. 2004

[a] - Data from presnt study reported as median (range)
[b] - Krahn et al. 1997. from Steller Sea Lion Recovery Investigations in Alaska 1995-1996
[c] - Concentrations reported in µg/g wet weight

Table 2.2. Steller sea lion (male) sample tissues obtained for PCB analyses

Sample ID	**Location**	**Tissues**			**Date**[a]
		Blubber	Liver	Kidney	
SSL-2 (1)	Tatitlek (PWS)		✓	✓	2/2/2001
SSL-2 (5)	Tatitlek (PWS)	✓	✓	✓	10/28/2000
SSL-1	Tatitlek (PWS)		✓	✓	2/2/2001
SSL-3	Tatitlek (PWS)	✓	✓	✓	10/28/2000
SSL-4	Tatitlek (PWS)	✓	✓	✓	10/28/2000
SNPSLS9802	St. Paul (BS)	✓	✓	✓	10/14/1998
SNPSLS9901	St. Paul (BS)	✓	✓	✓	4/29/1999
SNPSLS9808	St. Paul (BS)	✓	✓	✓	Fall-98
SNPSLS2000-04	St. Paul (BS)	✓	✓	✓	4/25/2000
Karin (SP-01-01-EJ)	St. Paul (BS)	✓	✓		3/23/2001
SP-01-00-EJ	St. Paul (BS)	✓			5/31/2000

PWS = Prince William Sound ; BS = Bering Sea

[a] month-day-year

Table 2.3. Steller sea lion placenta samples for PCB analyses

Sample ID	**Location**	**Date**[a]
Placenta-1	N. Ugamak	no date
Placenta-2	Unalaska	6/21/2000
Placenta-3	Unalaska	6/21/2000
Placenta-4	Japan	6/27/2001
Placenta-5	Seal Rock	7/6/2001
Placenta-6	Seal Rock	5/22/2002
Placenta-7	Seal Rock	5/22/2002
Placenta-8	Seal Rock	5/22/2002
Placenta-9	Seal Rock	5/22/2002

[a] month/day/year

Table 2.4. Concentrations of the sum PCBs (ΣPCBs) in the Steller sea lion tissues (male and female)

Tissue		Lipids (%)	sum PCBs concentrations (μg/g)			
			dw [a]		lw [b]	
			median (range)	iq R [c]	median (range)	iq R [c]
Blubber	n = 9	46	0.8 (0.4-1.5)	0.7	2.3 (1.1-2.9)	1.1
Liver	n = 10	6	0.2 (0.15-0.3)	0.06	3.7 (2.6 - 7.9)	1.8
Kidney	n = 9	5	0.06 (0.001-0.1)	0.03	1.3 (0.14 - 2.2)	1.1
			dw[a]		lw[b]	
			median (range)	iq R [c]	median (range)	iq R [c]
Placenta	n= 9	5	0.04 (0.02-0.1)	0.04	1.2 (0.4 - 6.6)	0.6

[a] dw = dry weight ; [b] lw = lipid weight ; [c] iq R = interquartile range

Table 2.5. Average toxic equivalents (TEQs) of the four detected dioxin-like PCB congeners in different tissues of Steller sea lions (pg/g lw)

Congener	**81**	**105**	**118**	**157**	**Sum average**
TEFs [a]	**0.0001**	**0.0001**	**0.0001**	**0.0005**	**TEQs**
Blubber	0.3	4.0	17.2	nd [b]	22
Liver	nd [b]	5.7	1.7	42.1	49
Kidney	nd [b]	1.9	6.7	nd [b]	9
Placenta	0.8	0.8	17.7	nd [b]	19

[a] - Van den Berg et al., (1998) ; [b] nd - not detected

Figure 2.1. Map of the northern Pacific Ocean showing locations of where Steller sea lion samples (male and female) were collected

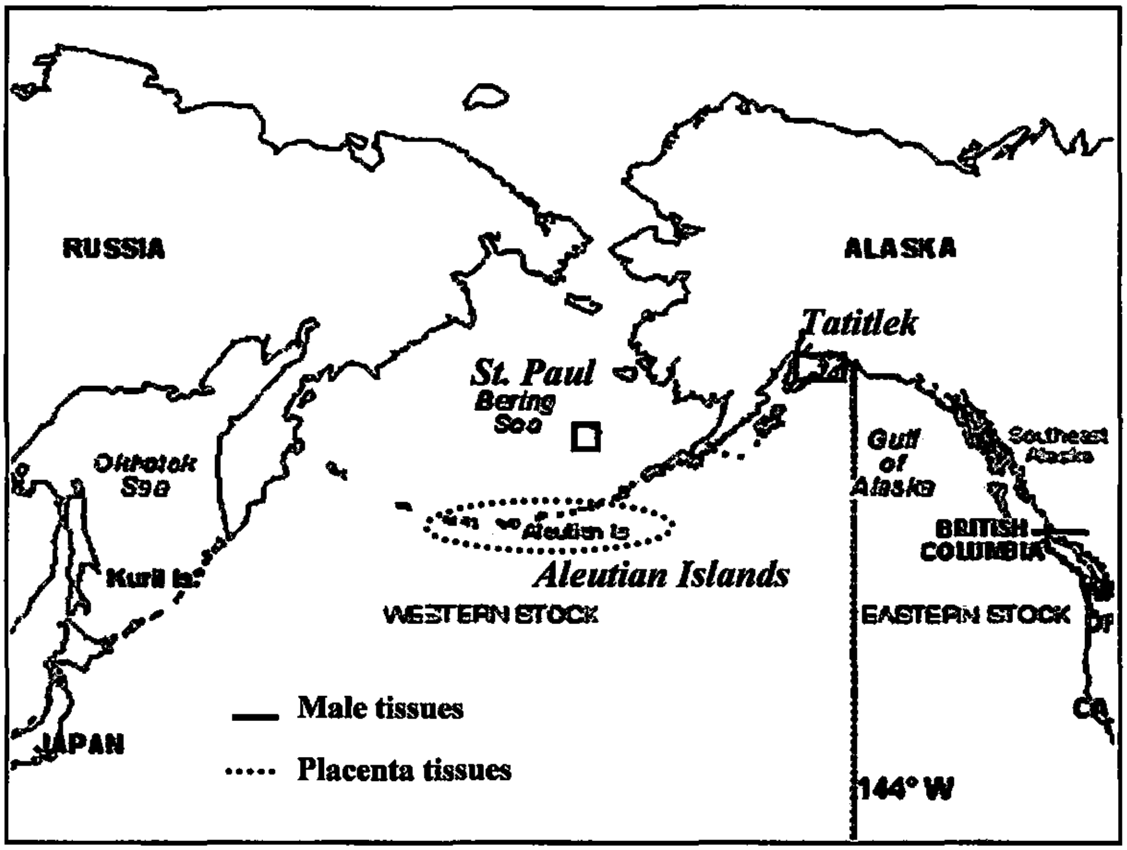


Figure 2.2. Boxplot of median, quartiles and single data points of PCB (µg/g *lw*) concentrations for male Steller sea lion tissues from two locations

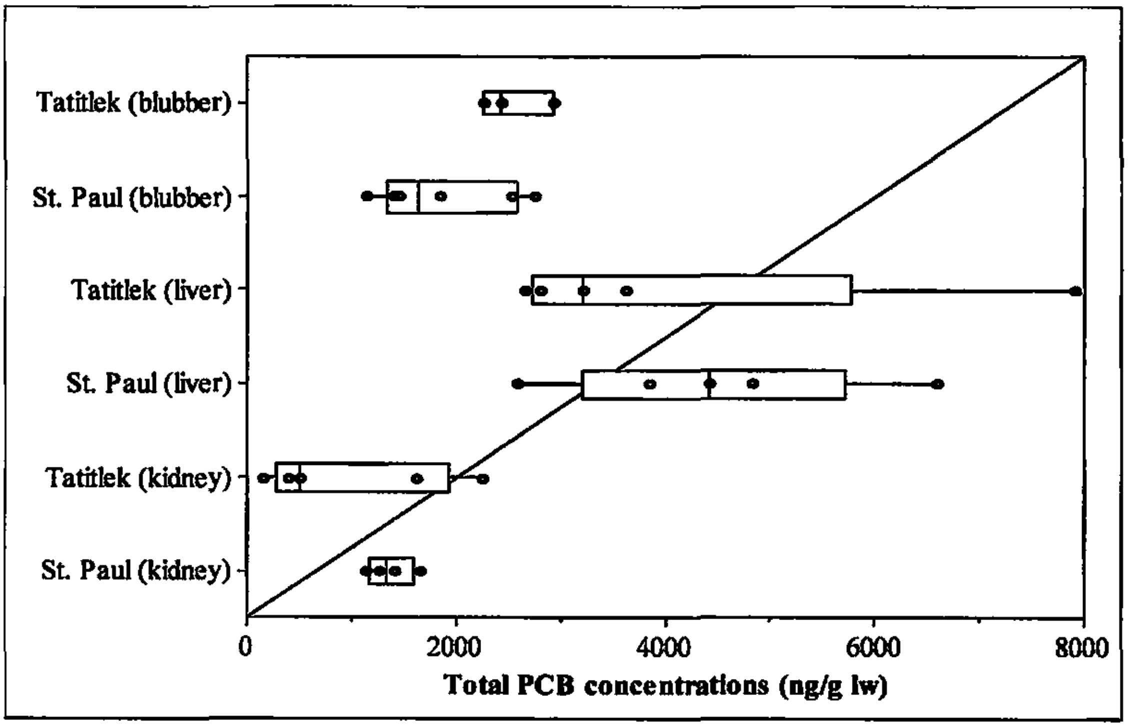


Figure 2.3. Taxonomy flowchart of selected Otariidae and Phocidae genus and species

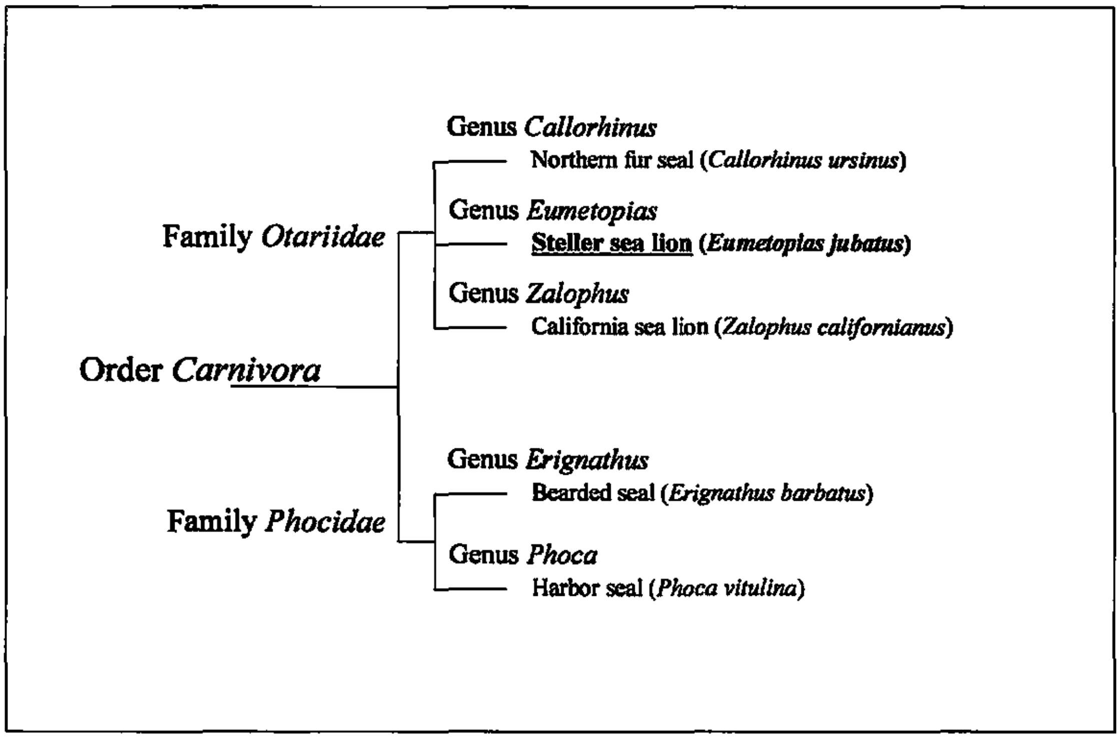


Figure 2.4. Median concentrations and ranges (µg/g *lw*) of total PCBs in blubber, liver and kidney tissue of Steller sea lions from Tatitlek (PWS) and St. Paul (BS)

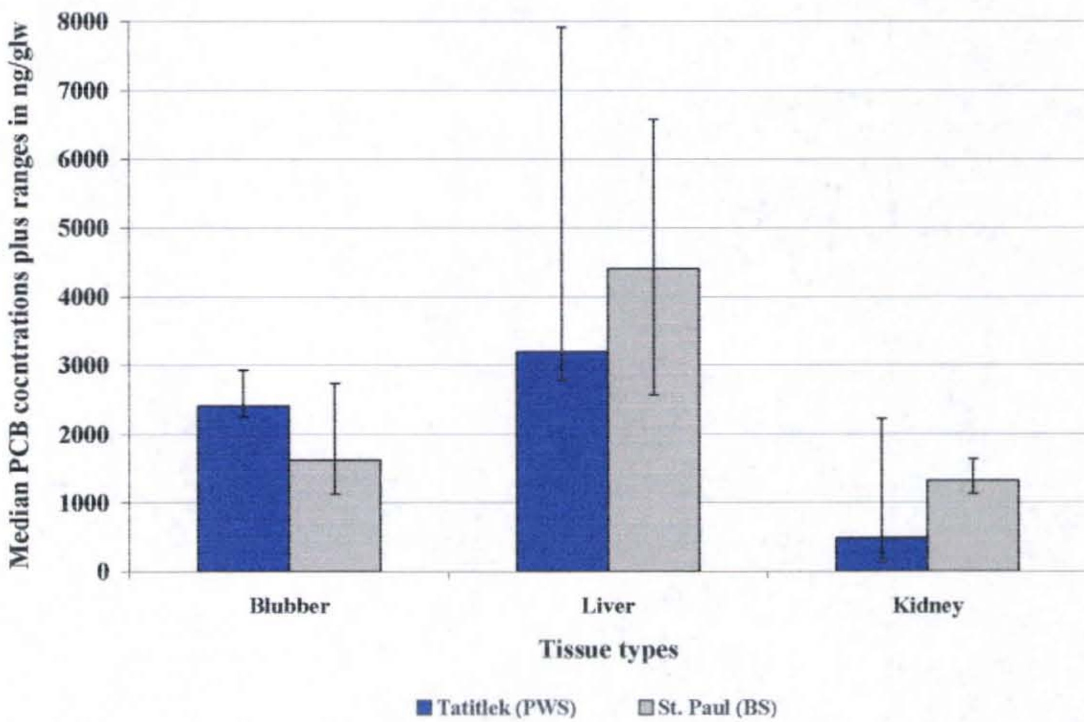


Figure 2.5. Composition of PCB homologues (average %) in Steller sea lion tissues and in Aroclors 1242, 1254 and 1260. The blubber, kidney and liver were from males and placenta from females

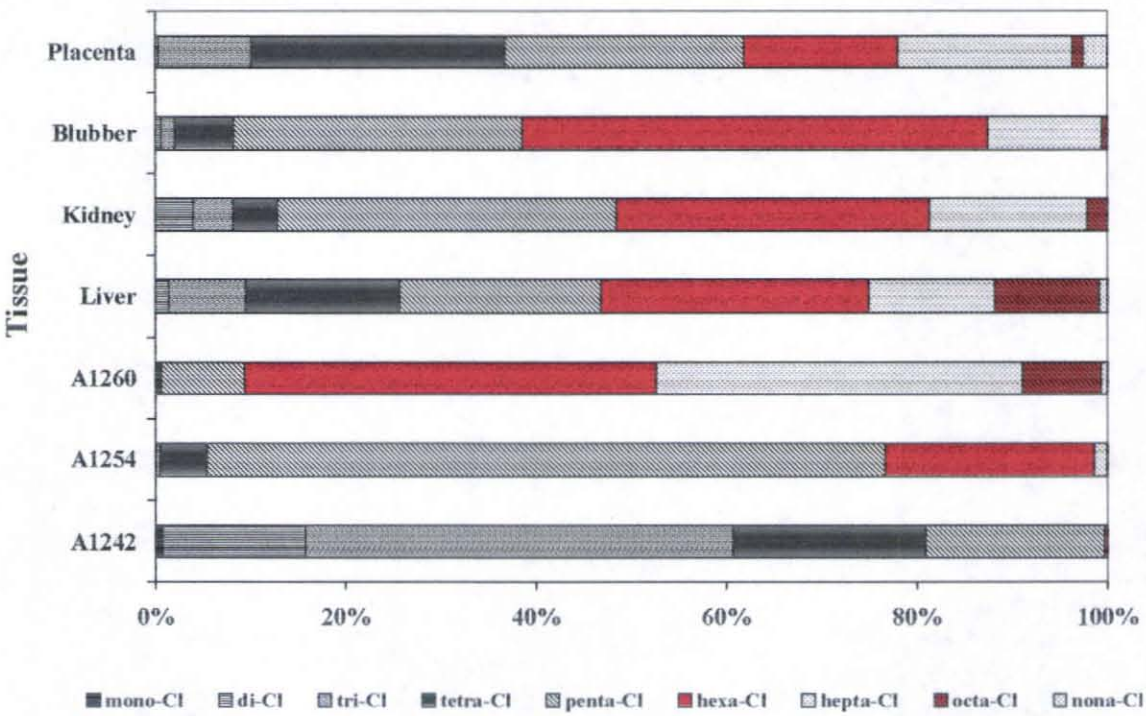


Figure 2.6. PCB congener profiles (average) in blubber, liver, kidney and placenta tissues from Steller sea lion (ng/g lw)

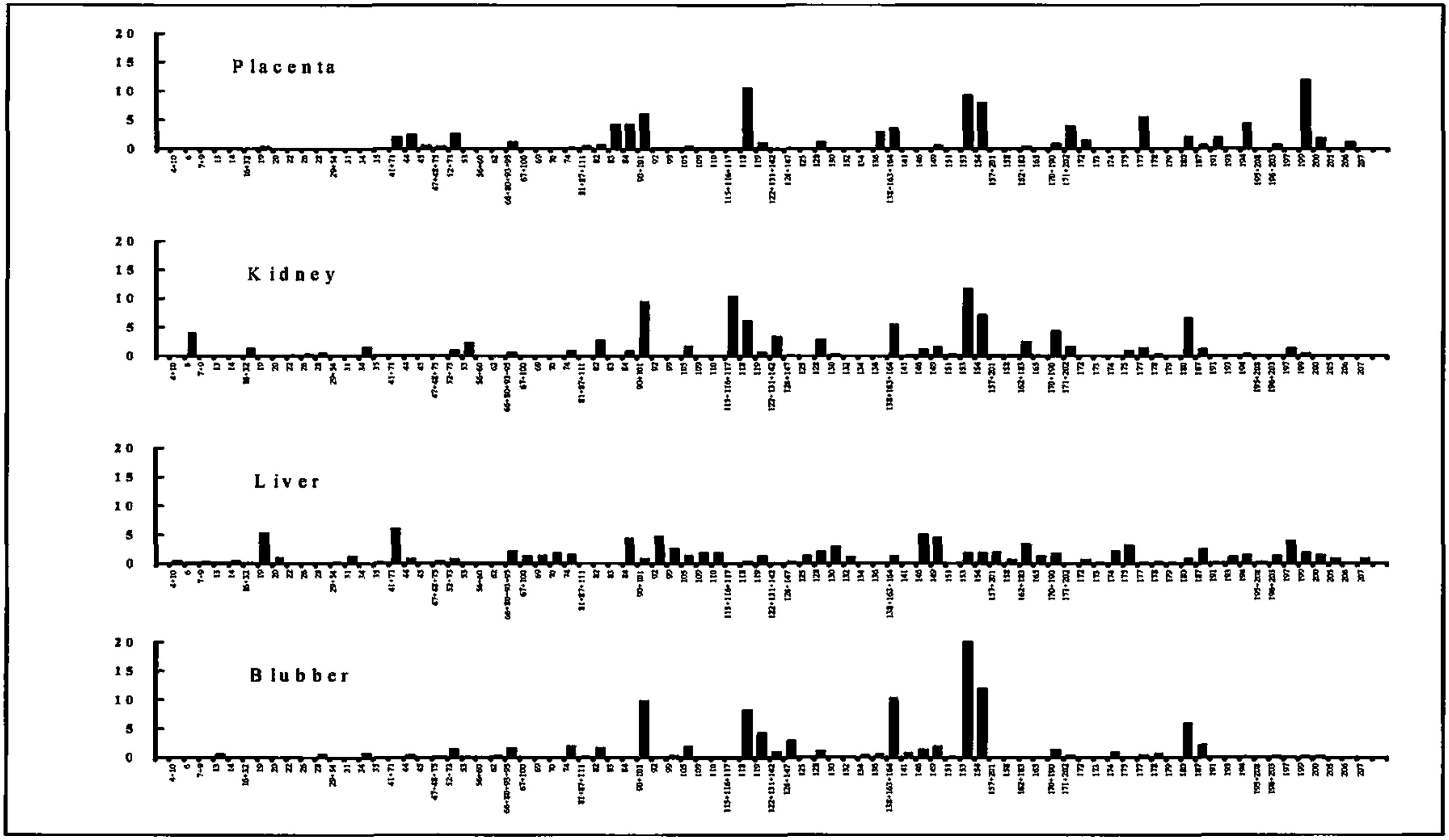
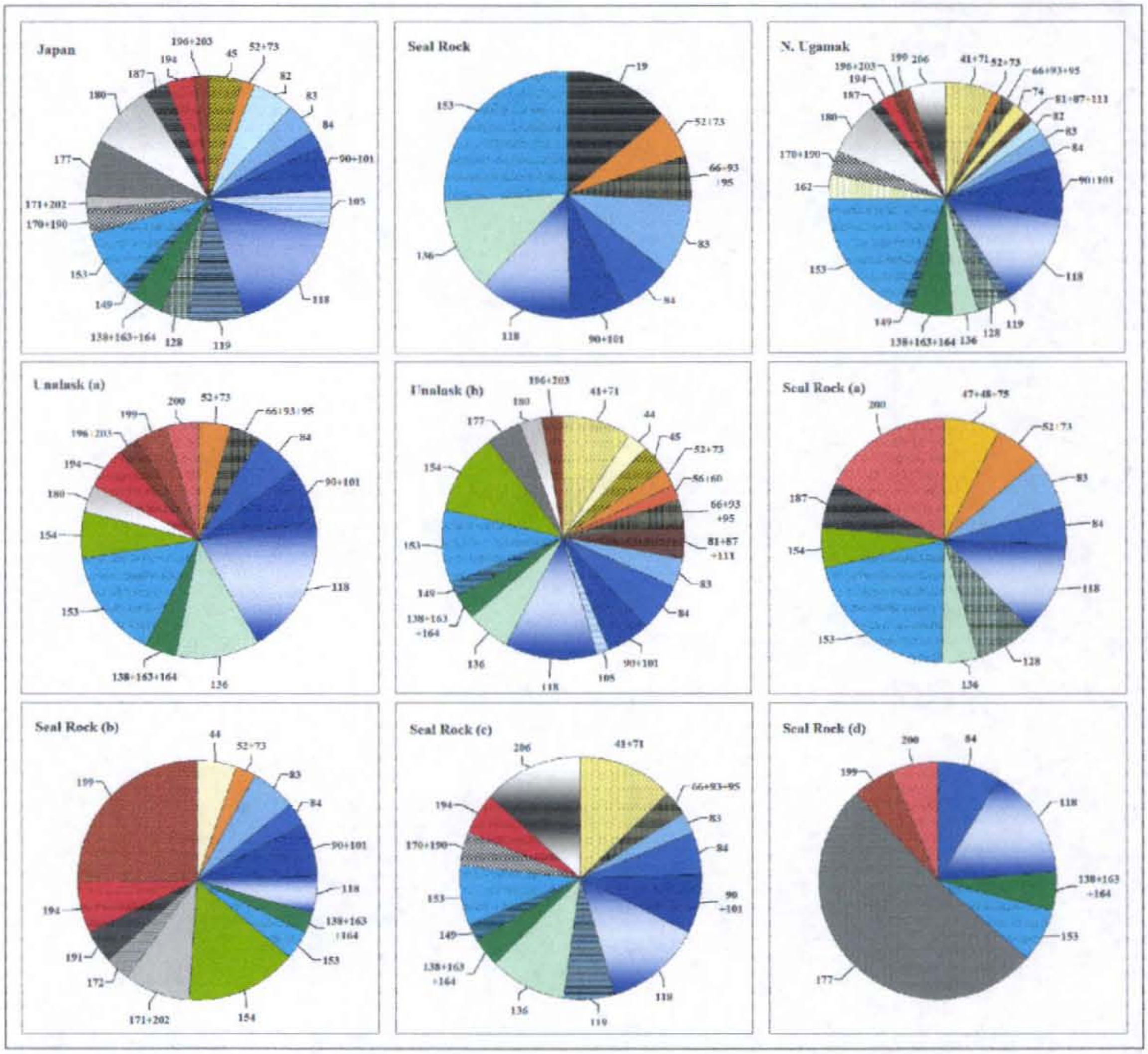


Figure 2.7. Profiles of 35 PCB congeners in nine Steller sea lion placenta samples



CHAPTER 3

SPATIAL AND TEMPORAL STUDY OF PERSISTENT ORGANIC POLLUTANTS IN STELLER SEA LION (*EUMETOPIAS JUBATUS*) IN THE NORTHERN PACIFIC OCEAN

Kathrin Hülck, Shannon Atkinson, Qing X. Li

3.1. Abstract

Environmental pollution is of increasing concern over its effects on environmental and human health. Persistent organic pollutants (POPs) can be detected in any environment around the world. None of these compounds occurs as a natural substance. Among POPs, two of the most thoroughly studied groups are polychlorinated biphenyls (PCBs) and organochlorine pesticides (OCPs). PCBs and OCPs bioaccumulate to high concentrations in body tissues of marine mammal especially in the Artic. Steller sea lions are endemic to the Northern Pacific Ocean and their western stock population has shown a decline of almost 80% since the 1950s. Today the number of Steller sea lions is only slowly recovering. Chronic exposure to PCBs and DDTs is among the suspected causes for their once drastic species decline. Limited data on POPs exposure and contamination in Steller sea lions are available. In this study, tissue samples for Tatitlek (Prince William Sound), St. Paul Island (Bering Sea) and Olutorsky Gulf (Russia) were used for a spatial and temporal comparison of Pops concentrations. In the spatial comparison, no significant differences in concentrations of PCBs were found between these two locations. The average concentrations of PCBs in blubber and liver were

below the immunotoxic threshold for blubber and physiological toxic threshold for liver. The TEQ values for individual animals were more than one order of magnitude below the theoretical threshold for toxic effects.

The temporal comparison of concentrations of different POPs (PCBs, HCHs, DDTs and HCB) in blubber and liver samples from 1976-1981 (Lee *et al.*, 1996) showed an overall trend of concentrations decreasing immensely (about one order of magnitude) over the last 20 years.

3.2. Introduction

Polychlorinated biphenyls (PCBs) and organochlorine pesticides (OCPs) are ubiquitous in the environment. The use as well as production of PCBs and most OCPs were restricted worldwide and banned in North America and Europe in the 1970s and 1980s. PCBs consist of 209 individual congeners, which differ only by the number of chlorine atoms attached to the basic biphenyl structure at different positions (Ballschmitter and Zell 1980). Between 1929 and 1977 approximately 700,000 tons of pure PCBs were produced in the United States. OCPs include a large number of pesticides such as the most known DDT (dichloro-diphenyl-trichloroethane) and its degradation products DDE (1,1-dichloro-2,2-bis(p-dichloro-diphenyl)ethylene) and DDD (1,1-dichloro-2,2-bis(p-chlorophenyl)ethane). Others OCPs are lindane (α-, β-, γ-, δ-HCH isomers), heptachlor, and hexachlorobenzene (HCB). All of these compounds have been used extensively in the past, but like PCBs have been restricted in many parts of the world. POPs are very stable and irrepealably enter the environment. This structurally diverse group of compounds has been transported to the Arctic by long range atmospheric and oceanic vectors. Local point sources or emissions are another way of entry. For example, PCBs were used in technical mixtures known as Aroclors (Kimbrough 1989). According to their specific applications different mixtures with higher or lower chlorinated congeners were composed. For example, Arcolor 1242 was used in transformers, hydraulic fluids, plasticizer, carbonless paper, adhesives and wax extenders (IARC 1979; Safe, 1984). POPs adsorb strongly to atmospheric particles, soil, and sediments (Muir *et al.*, 1999). They undergo long-range transport (LRT) via "cold-condensation" (Wania and Mackay, 1993). As latitudinal and vertical temperature changes, POPs undergo equilibrium partitioning between gas and solid (surface) phases (Wania and Mackay 1996, Wania 1999). Thus, POPs can be found in the most remote areas of the world, e.g., Arctic and Antarctic regions. POPs usually have a octanol-water partitioning coefficient (K_{ow}) value >1. This partitioning coefficient is used to determine the partitioning between POPs in water and body fat of animals (Veith *et al.*, 1979). POPs enter the food chain, e.g. through benthic organisms, which filter or directly ingest contaminated sediments or direct absorption through the skin. Bioaccumulation of POPs in animal tissues occurs mainly via the food chain, also known as biomagnification (Hornshaw *et al.,* 1983; Rubinstein *et al.*, 1984; Rasmussen *et al.*, 1990; Barron 1995; Harding *et al.,* 1997; Borgå et al., 2001; Muir *et al.*, 2003). Direct correlations between body fat content and contaminant concentrations have been observed for many animal species (Holden, 1973). Due to the high lipid content and length of the food chain (biomagnification potential), POPs have been accumulated at relatively high concentrations in top predatory marine mammal species (Bruhn *et al.*, 1995; Jenssen *et al.*. 1996; Ross 2004). The large amount of body fats originate in their biology and marine environment to provide buoyancy, streamlining and insulation, making them most vulnerable to bioaccumulation of POPs. Bioaccumulation of POPs in blubber of various marine mammals has been shown (Becker *et al.*, 2000; Helm *et al.*, 2002, Hall and Kalantzi, 2003; Hobbs and Muir, 2003). Toxic effects of POPs on marine mammals and other animals include reproduction failure, weakened immune systems, hormone disruption or chronic health disorders (Reijnders, 1986; Law *et al.*, 1989; Olsson *et al.*, 1994, Ross 1996).

PCBs exert toxic effect similar to 2,3,7,8- tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD), the most potent man-made chemical (Castello *et al.*, 1997; Van den Berg *et al.*, 1998; Geyer *et al.*, 2000). A Toxic Equivalent (TEQs) scheme was established by Van den Berg *et al.* (1998) and used to report the toxicity of chemicals relative to the toxicity of 2,3,7,8-TCDD. TEQs can be reported separately for Each of the 12 dioxin-like PCB congeners can be described by TEQs or combined as ΣPCB-TEQs according to the mixtures found in individual animal tissues. TEQs offer a method of reporting toxicity normalized to that of 2,3,7,8-TCDD, rather than reporting the total number of grams of a mixture of variable toxic compounds. Normalizing PCB toxicity is an important tool for ecotoxicological risk assessment and management.

Since the 1970s marine mammals have been extensively studied for POPs. These studies were continued and extended to various tissues for pharmacokinetic studies as well as different species (Aguilar and Borrell 1991, Severinsen and Skarre 2000, Wang *et al.*, 2006). Today marine mammals are among the key animals in contaminant studies because (1) of their longevity and principal position as top-predators in many food chains around the world; (2) their body fat shows the highest overall concentrations of contaminants in the body; (3) marine mammal populations such as pinnipeds (seals, sea lion and walruses) are declining; (4) marine mammals are hunted for subsistence, making humans the end bioaccumulator of POPs; and (5) knowledge of distribution and pharmacokinetics of POPs can help accurate assessment and proper management of human health risks.

One of the top predators in the Northern Pacific Ocean is the Steller sea lion (*Eumetopias jubatus*). These Otariidae can be found in the Northern Pacific from all areas of the Bering Sea (Alaska, Aleutian Islands and Russia) down to the northern part of California. Protected in 1990 under the Endangered Species act, the western population of this species has still been declining over the last decades. The western stock has declined by about 85% up to now, since the mid to late 1980s. Additionally from the remaining numbers nearly another 63% declined from 1990 to 2002 (Marine Mammal Commission, 2003). Two hypotheses for this decline currently under investigation include changes of major prey species and changes in habitat and long-term exposure to POPs.

Using data from almost 20 years ago, Lee *et al.* (1996) were among the first to report total PCB and OCP concentrations in Steller sea lion tissues in the Northern Pacific Ocean. POPs have been used to some extent in the populated areas of the Northern Pacific Ocean (North America, Russia, Japan etc.), but the question of point source emission vs. long-range transport remains. Weathered POP profiles, observed in biota exist throughout the Northern Pacific Ocean, are not comparable to those of standard technical mixtures or formulations. In this study, samples of Steller sea lions from the Russian region (Olutorsky Gulf) were available and compared to those of the Pribilof Islands (St. Paul, Bering Sea) and Prince William Sound (Tatitlek, Alaska) (Figure 3.1). With Steller sea lion population numbers still declining in the western region, the question of how overall concentrations of POPs may negatively impact these top predators remains. In addition, have restrictions in the production and use of POPs, as well as changes in their transport and breakdown lead to an overall lower concentration worldwide?

This chapter describes POP concentrations and profiles in the blubber and liver samples of Steller sea lion from three distinct regions of the Northern Pacific Ocean. PCB congener specific profiles and TEQs were established and profile similarities and toxicity evaluated. POP concentrations were compared to historical data of POPs in Steller sea lion from similar regions from almost 20 years ago (Lee *et al.*, 1996). Such comparisons will help to define a possible cause of Steller sea lion population decline and provide a picture of Steller sea lions exposure to POP and the bioaccumulation of these compounds in this species.

3.3. Materials and Methods

3.3.1 Collection of Samples

Steller sea lion blubber and liver samples were obtained from subsistence hunts from October 2000 to April 2002 for Tatitlek (PWS) and from October 1998 to November 2001 for St. Paul Island (BS). Tissues of ale Steller sea lion blubber from Olutorsky Gulf (Russia) were collected by local Fisherman from animals entangled in nets (October- December 2002). All adipose tissue (blubber, n=28), and liver (n=16) sub samples were collected, wrapped in aluminum foil, and immediately frozen at –20 °C until they were processed (Table 3.1). Additional PCB data presented in chapter 2 for Steller sea lion male blubber and liver tissues (Table 3.2) were used in the overall location comparison of total PCB concentration.

3.3.2. Chemical Analysis

Altogether 28 PCBs congeners and 12 OCPs were analyzed in triplicates or as otherwise stated. The 28 PCBs congeners were selected for analysis based on the recommendations by USEPA, WHO and NOAA (NOAA, 1989). They included 12 dioxin-like PCBs and 16 of the most frequently detected and most persistent PCB congeners. Twelve OCPs were chosen due to the similarity of their chemical properties to those of PCBs (ATSDR 1998, 2000, 2002, 2005). Pork lard and chicken liver samples were used as surrogate matrices to determine recoveries according to the method of Wang *et al.* (2006) with other slight modifications. Tables 3.3., 3.4. and 3.5. show the recoveries, relative standard deviations (RSD) and method detection limits (MDL) for all the PCBs and OCPs in surrogate lard and chicken liver. Tissue Samples were sawed, cut, homogenized and lyophilized (Freeze Dry System Model 77530, Labconco) for 2 days. One gram (dry weight, *dw*) of lard and 2 g of liver tissues were each individually placed into supercritical fluid extractor cells. Approximately 1 inch Ottawa sand (baked) was added before and after the tissue samples. Stable isotope internal method standards ^{13}C-PCB 28/123/170 (100 pg/μl each) were added to each sample. These internal standards were selected to spike surrogate and actual samples to assure the data quality.

Samples were extracted using a SFX 220 (supercritical fluid extraction system; Teledyne Isco, Inc., Lincoln, Nebraska) 10-min static extraction followed by a 40-min dynamic extraction at 6000 psi and 140°C. The extracts were collected in 25 ml glass tubes containing hexane (Optima, Fisher Scientific). The extractor lines were cleaned between extractions by running hexane/acetone (1:1, v:v) for 6 min (Optima, Fisher Scientific). Sample extracts were transferred into volumetric test-tubes (15 ml). Glassware was rinsed 3 times after each transfer with ~2 ml of the corresponding solvent and the rinsates were added to the extracts. Extracts were concentrated to 8 ml under a gentle nitrogen stream. One milliliter of each extract was withdrawn and added to pre-weighted alumina cups for gravimetric determination of lipid weight. Remaining extracts were once again concentrated to ~2 ml under a gentle nitrogen stream. For extract clean-up and lipid removal, extracts were transferred to glass columns containing 10 g of 2% deactivated florisil gel (activated at 120°C overnight, 2% of distilled deionized water added). The column was eluted with 60 ml of (85:15, v:v) hexane:methylene chloride (methylene chloride HPLC GC resolve, Fisher Scientific). The eluates (60 ml) were collected and concentrated under vacuum on a rotatory evaporator at 40°C. The florisil-cleaned eluates were transferred to glass columns containing 8 g of 40% sulfuric-acid/silica (silica activated in oven at 120°C overnight, 40% sulfuric-acid was added). The column was eluted with 60 ml of hexane and the eluates collected in round bottom flasks. The silica-cleaned eluates were concentrated to 0.5 ml under a gentle nitrogen-stream. An amount of ~ 0.2 ml isooctane (Optima, Fisher Scientific) and 100 pg/µl of ^{13}C-PCB 169 (as stable isotope internal GC/MS instrument standard) were added. The eluates were concentrated to 100 µl and were transferred into GC-vials with 100 µl conical inserts. GC-vials were labeled and stored at –20°C until GC-MS analysis. C-WNN Standard and a mixture of 12 OCPs each at 10, 100, 500, 1000, 1500, and 2000 ng/ml were prepared and run on the GC-MS with every set of samples to establish the relative response factors (RRFs) for each of the target (native) analytes. These are defined as follows:

$$RRF = (PA_{native}/C_{native})/(PA_{C13}/C_{C13}) \qquad \text{(Equation 3.1)}$$

(PA - Peak Area; C - Concentration)

Blank hexane and isooctane were run for quality assurance purposes. Samples were analyzed on a Varian CP-3800 GC ECD and Varian Saturn 2000 ion trap MS in split mode (1/9, ECD/MS). Chromatographic separation was performed on a ZB-1 capillary column (60 m x 0.25 mm I.D.). The column oven temperature was programmed from 120°C to 275°C (hold 10 min.) at a rate of 2°C /min. Injector and detector temperatures were held at 280°C and 330°C, respectively. Helium with a flow rate of 2.0 ml/min was used as the carrier gas and nitrogen used as the make-up gas. Concentrations reported in this study were not corrected for recoveries of stable isotope internal method standards. Concentration calculation are defined as follows:

$$C_{sample} = PA_{native}/RRF \bullet (PA_{standard}/100) \qquad \text{(Equation 3.2)}$$

($PA_{standard}$ - ^{13}C- PCB average Peak area)

3.3.3. Statistical Analysis

Concentrations of POPs in blubber and liver tissues of Steller sea lion (male and females) from the three distinct locations were analyzed statistically using JMP 5.1 software (SAS Institute, Inc.). These tissue samples could not be defined as being normal distributed due to the small sample sizes of all tissue types. Thus, for PCB and OCP concentration visualization and comparisons, median, interquartile range and concentration range (min-max) were chosen as well as non-parametric Wilcoxon-Mann-Whitney test and the Kruskal-Wallis test for the data analysis. The Wilcoxon-Mann-Whitney test (also known as the Mann-Whitney *U* test) is a non-parametric test for assessing whether the medians between two samples of observations are the same. The Kruskal-Wallis test is also non-parametric and used to compare three or more independent groups of samples (it can be used as a non-parametric alternative to the one-way ANOVA). The level of significance for both tests was set to $p<0.05$. Box-plot graphs were generated with Minitlab®Release14.20 (Minitlab, Inc.).

3.4. Results and Discussion

3.4.1. Spatial comparison and gender comparison

The total PCB concentrations in the Steller sea lion tissues were reported as median concentrations on a lipid weight (*lw*) basis. The total PCB (28 PCB congeners) concentrations for blubber from male Steller sea lion from Tatitlek (PWS), St. Paul Island (BS), and Olutorsky Gulf (R) were 956, 2022 and 1283 ng/g *lw*, respectively (Table 3.6). The blubber tissues from St. Paul Island showed the highest median total PCB concentrations (Figure 3.2). Statistical analyses showed that total PCBs in male blubber tissues collected from St. Paul and Olutorsky Gulf were significant higher then those from Tatitlek. Total PCB concentrations in blubber from Olutorsky Gulf and St Paul Island showed no significant difference.

Total PCB concentrations in the liver tissues of male Steller sea lion were 238 ng/g (Tatitlek) and 755 ng/g *lw* (St. Paul). The liver tissues from Olutorsky Gulf were not available for analysis. Total PCB concentrations in female Steller sea lion tissues (Tatitlek only) were 527 ng/g for blubber and 173 ng/g *lw* for liver (Table 3.6). The statistical comparison between female and male blubber and liver samples from the one common location (Tatitlek, (PWS)) showed no significant differences (Table 3.6, Figure 3.3).

Varanasi *et al.* (1992) were amongst the first to report PCB and OCP data for Steller sea lion. Compared to the present study, concentrations of total PCBs and DDTs were about one magnitude higher for blubber tissues (Table 3.7), but similar for liver tissues. A temporal comparison of total PCB concentrations in Steller sea lion from the present study with data from Lee *et al.* (1996) will be discussed in this chapter, section 2.3. Literature data (Table 3.7) from Krahn *et al.*, (2001) and those from chapter 2 showed total PCB concentrations for blubber tissue in the same range, but in the present study, total PCB concentrations in the liver were lower. This could be due to differences in analytical methods as well as the definition for "total PCB". Overall, 67 PCB congeners were detected in the study described in chapter 2, whereas in the present study attention was paid to 28 PCBs. These were chosen because they are known to include the most persistent PCB congeners 44, 52, 138, 153, 170, 180, and 187, as well as the 12 dioxin-like PCBs (77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169, and 189).

Because the habitats of both Steller sea lion and Northern fur seals overlap at St. Paul Island, comparisons between these species can be useful. Concentrations of PCBs in Northern fur seal blubber from the Pribilof Islands (Loughlin *et al.*, 2002) were between 3032 ± 1074 ng/g *lw* (subadults from St. George) and 2400 ± 855 ng/g *lw* (subadults from St. Paul). These concentrations lie within the same range as those of the present study. Total PCB concentrations were recently reported by Wang *et al.* (2006) for Harbor seals from PWS and showed about one order of magnitude lower concentrations then those for Steller sea lion blubber and liver. It is noted that the Harbor seals (Wang *et al.*, 2006) share a similar habitat with the Steller sea lions in this study, but are in a different taxonomical family and occupy a lower trophic level in the Arctic food chain.

Among the 28 PCBs analyzed, the four dominant PCB congeners were 153, 138, 118 and 180 for male and female Steller sea lion blubber from Tatitlek (PWS) and males from St. Paul Island (Table 3.8). In comparison the four dominant PCB congeners in Olutorsky Gulf samples were 153, 138, 118 and 123 (Figure 3.4). PCB congeners 118 and 123 are among the dioxin-like PCB congeners. PCB 123 was detected in 8 out of 10 male Steller sea lion blubber samples from Olutorsky Gulf but detected only in 2 of the 7 male blubber samples from St. Paul Island, 1 of the 5 female blubber samples from Tatitlek and in none of the male Steller sea lion blubber samples from Tatitlek (Table 3.8).

The four dominant PCB congeners in liver samples from St. Paul and Tatitlek were 153, 138, 118 and 180 (Figure 3.4). PCB congener 123 was not detected in any of the liver samples. PCB congeners 153, 138, 118 and 180 are among the most recalcitrant congeners and are usually found as the dominant PCB congeners in various tissues of marine mammals around the world (Boon *et al.*, 1997; Kleivange *et al.*, 1995; Kleivange, 2000; Kucklick *et al.*, 2002; Harju *et al*, 2003; Muir *et al.*, 2003; Storelli and Marcotrigiano, 2003). The median PCBs concentrations in liver of 0.3 μg/g *lw* for male Steller sea lion from Tatitlek and 0.8 μg/g *lw* from those from St. Paul were below the physiological toxic threshold of 6.6-11 μg/g *lw* in liver of marine mammals (Kannan *et al.*, 2000). This also applies for PCBs concentrations of 0.2 μg/g *lw* in female Steller sea lion liver from Tatitlek. It is noted that the total PCBs concentrations in all the blubber samples in this study (0.9, 2.0 and 1.3 μg/g *lw* for males and 0.5 μg/g *lw* for females from the various sample areas) were below the immunotoxic threshold of 17 μg/g (Kannan *et al.*, 2000).

Comparison of chlorinated PCBs homologue profiles in the tissues with those of Aroclor technical mixtures can be used to elucidate possible source of PCBs. Homologue profiles of the 28 target PCBs in different Aroclor mixtures were compiled for direct comparison (Figure 3.5). Hexachlorobiphenyls were the prevalent homologues in blubber and liver tissues of Steller sea lion (Figure 3.5) followed by pentachlorobiphenyl homologues. This homologue profile was observed in blubber and liver tissues regardless of gender or location. Compared to seven Aroclor mixtures, tissue homologue profiles most closely resembled those of Aroclor 1254a and 1254b (Monsanto Chemical Company, IL). Aroclor 1254a and 1254b were amongst the most widely used Acolors produced by Monsanto in the US (IARC 1979). It is evident that pentachlorobiphenyls are the mayor homologue in the Aroclor mixtures 1254a and 1254b followed by hexachlorobiphenyls. PCB profiles in tissues were apparently weathered. A lower persistence of pentachlorobiphenyls in the environment relative to hexachlorobiphenyls may explain the differences between tissue homologue profiles and Aroclor 1254a and 1254b profiles.

Twelve OCPs were analyzed in the blubber of male and female Steller sea lion from the Northern Pacific Ocean (Table 3.9). Three OCPs, α-HCH, β-HCH, and γ-HCH were detected and combined as ΣHCHs for data comparison. None of the samples was found to contain δ-HCH. Total concentrations of HCHs in blubber were 146, 424 and 227 ng/g *lw* for Tatitlek, St. Paul Island and Olutorsky Gulf, respectively (Table 3.9). Concentrations of α-HCH in male blubber samples were not significantly different between the three locations. Concentrations of β-HCH showed significant differences between all the locations, with the highest concentrations observed in samples from St. Paul Island followed by Olutorsky Gulf and Tatitlek. Significant differences in concentrations, of γ-HCH existed between samples from Olutorsky Gulf and the other two locations. ΣHCHs showed significant differences for all three locations, with concentrations highest at St. Paul Island followed by Olutorsky Gulf and Tatitlek (Figure 3.6).

Female Steller sea lion blubber from animals collected in Tatitlek contained 56 ng/g *lw* of total ΣHCHs for Tatitlek (Table 3.9). Significant differences were not observed for isomers α-HCH, β-HCH, γ-HCH as well as ΣHCHs between the male and female Steller sea lion tissues at Tatitlek (PWS).

HCB and heptachlor were also detected in male animals at concentrations of 2, 4, 1 ng/g *lw* and 19, 21 and 9 ng/g *lw* at Tatitlek, St. Paul Island and Olutorsky Gulf, respectively. HCB concentrations in samples from St. Paul Island were significantly higher that those from Tatitlek and those from Olutorsky Gulf. Female animals showed concentrations of 3 ng/g *lw* for HCB and 14 ng/g for heptachlor. There were no significant differences between male and female blubber tissues from Tatitlek (PWS) in regards to either HCB or heptachlor (Figure 3.6).

Concentrations of o,p-DDE, o,p-DDD and o,p-DDT were pooled to yield total Σo,p-DDTs. These totals were 39, 28 and 79 ng/g *lw* in male Steller sea lion from Tatitlek, St. Paul Island and Olutorsky Gulf, respectively, and 9 ng/g *lw* in female Steller sea lion blubber from Tatitlek (Table 3.9). Concentrations of o,p-DDE in samples from Olutorsky Gulf were significantly higher than those from St. Paul Island. No significant difference was observed for concentrations of o,p-DDT and Σo,p-DDTs among the three locations. Concentrations of p,p-DDT p,p-DDE, and p,p-DDD were combined as Σp,p-DDTs (Table 3.9). The Σp,p,-DDTs were 952, 3023 and 1448 ng/g *lw* for samples from Tatitlek, St. Paul Island and Olutorsky Gulf, respectively. Females from Tatitlek had 489 ng/g *lw* Σp,p,-DDT (Table 3.9). The concentrations of the metabolites p,p,-DDE and p,p,-DDD, the parent p,p,-DDT concentrations and Σp,p-DDTs were all significantly higher in samples from St. Paul Island than in those from Tatitlek and Olutorsky Gulf.

The Σo,p-DDTs and Σp,p-DDTs groups both showed large differences between male and female blubber tissues at Tatitlek (Table 3.9). Concentrations of o,p-DDD, o,p-DDT and Σo,p-DDTs, except for o,p-DDE, were significantly higher in the male Steller sea lions than in females from Tatitlek (Table 3.9). Median concentrations of Σp,p-DDTs and their single isomers in males at Tatitlek (PWS) were higher than in females, but not significantly. Like other female pinnipeds, Steller sea lion females are known to transfer part of their POPs body-burden to their pups during pregnancy and lactation (Lee *et al.*, 1996) while male Steller sea lions accumulate POPs throughout their life span.

Krahn *et al.*, (2001), recently published data of DDTs in Steller sea lion tissues (Table 3.7). Median concentrations were 5000 ± 600 ng/g *lw* for female and male juveniles from Southeast Alaska. These concentrations are in a similar range as Σp,p-DDTs in male Steller sea lion from St. Paul Island, but above those from Tatitlek (Prince William Sound) and Olutorsky Gulf (Russia). Further comparisons of OCPs in Steller sea lion with data of Lee *et al.*, (1996) were discussed in this chapter, section 2.3. (Temporal comparison).

It was found that DDTs accumulated in other marine mammals such as Northern fur seals, harbor seals and ringed seals in Alaskan waters (Becker *et al.*, 2000) at similar concentrations to those in the present study. Mean concentrations of DDTs (sum of o,p and p,p DDTs) in blubber of mature male harbor seals and grey seals from the St. Lawrence Estuary, Canada were 8590 ± 2670 ng/g *lw* and 4210 ng/g *lw* (Bernt *et al.*, 1999). Mature females showed mean concentrations (DDTs) of 3630 ± 615 ng/g *lw* in harbor seals and 1920 ± 926 ng/g *lw* in grey seals. The concentrations of DDTs in male and female harbor and grey seals were above those of the present study. These samples were from a location in eastern Canada within close continental proximity, unlike those from the remote St. Paul and Olutorsky Gulf. Data reported by Bernt *et al.* (1999) are more comparable to the high concentrations of DDTs (single or sum) found in other studies of ringed seals from the North Sea (Severinsen *et al.*, 2000) and the harbor seals from the northwestern Atlantic coast (Shaw *et al.*, 2005). These latter areas are all close to centers of extensive industrial and anthropogenic activity.

3.4.2. 2,3,7,8-TCDD toxic equivalents (TEQs)

Mammal-specific toxic equivalent factors (TEFs) (Van den Berg *et al.*, 1998) were used to calculate the toxic equivalents (TEQs) of the detected dioxin-like PCB congeners. These are defined as follows:

$$\text{PCB congener concentration (pg)} \times \text{TEF} = \text{TEQ (pg)} \qquad \text{(Equation 3.3)}$$

Of the existing 12 dioxin-like PCB congeners, 10 were detected in this study (Table 3.10). Not all of the 10 dioxin-like PCB congeners were found in each sample. PCB 81 was detected in only one male blubber tissue from Tatitlek. PCB 126 was found in only one male liver sample from Tatitlek. Congeners 77 and 156 were detected in some of the blubber and liver samples from different locations (Table 3.10). Dioxin-like congeners 123, 157 and 167 were detected only in the blubber samples of male and female Steller sea lion (all locations), but not in any liver samples. Nine dioxin-like PCBs were found (77, 81, 105, 114, 118, 123, 156, 157 and 167) in the blubber. PCBs 105, 118 and 167 were the only congeners detected in all male and female Steller sea lion blubber tissues regardless of the location. Congeners 105, 114, 118, 156 and 167 were the only four dioxin-like PCBs found in both the blubber and liver tissues (Table 3.10).

Six dioxin-like PCB congeners (105, 114, 118, 126, 156 and 167) were detected in the livers of male and female Steller sea lion (Table 3.10). Only congener 105 and 118 were detected in all the liver samples regardless of gender or location. The absence of 123, 157 and 167 in the liver tissue could be explained by active liver metabolism of PCBs (Watanabe *et al.*, 2000; White *et al.*, 2000) as compared to the blubber that is viewed as a storage organ (Severinsen *et al.*, 2000). The ΣPCB-TEQs for male Steller sea lion tissues from Tatitlek were 17 pg/g *lw* (blubber) and 40 pg/g *lw* (liver). For animal tissues collected at St. Paul, ΣPCB-TEQs were 78 pg/g *lw* (blubber) and 13 pg/g *lw* (liver). For female Steller sea lion, ΣPCB-TEQs were 18 pg/g *lw* for the blubber and 5 pg/g *lw* for the liver tissues. Olutorsky Gulf male Steller sea lion showed ΣPCB-TEQs concentration of 41 pg/g *lw* (Table 3.10). Mean ΣPCB-TEQs of Steller sea lions described in chapter 2 were 9, 49, and 22 pg/g *lw* in the kidney, liver, and blubber samples, respectively. Wang *et al.* (2006) showed comparable ΣPCB-TEQs for Harbor seal liver and blubber tissues (12-22.5 pg/g and 8.6-173 pg/g *lw*) from similar locations. The TEQ values for individual animals were more than one order of magnitude below the theoretical threshold for toxic effects (520 pg/g *lw*, Kannan *et al.*, 2000).

3.4.3. Temporal comparison

PCBs were analyzed in male and female blubber and liver tissues. Male samples were collected at Tatitlek (Prince William Sound), St. Paul Island (Bering Sea) and Olutorsky Gulf (Russia). Female blubber and livers were collected at Tatitlek (PWS) (Table 3.1). About 20 years ago samples (blubber and livers) were analyzed by Lee *et al.* (1996) for POPs including ΣPCBs, p,p-DDD, p,p-DDE, p,p-DDT, ΣDDTs, α-HCH, β-HCH, γ-HCH, ΣHCHs and HCB. Data of this study (samples collected 1998-2002; Table 3.1) were compared with those of Lee at al. (1996). Due to the occurrence of abiotic and biotic degradation in the environment and biota (Brown and Wagner, 1990; Lake *et al.*, 1992), and restrictions in their use, concentrations of these man-made pollutants should have declined over the past 20 years.

A comparison, however, has to be made cautiously. Sample collection and homogeneity (Aguilar *et al.*, 2002), preservation as well as analytical methods and data analysis have advanced and results from more recent studies are difficult to compare to those obtained with older methods. Furthermore, the analysis of sample tissues was carried out in different laboratories, which introduces another variation. Samples were collected from two or more locations in the Northern Pacific Ocean, but these also vary by study. Therefore, any comparison should be viewed more as an update and an overall picture of POPs pollution over time. Last, but not least, multiple variables exit among individual animals, e.g., age, gender, metabolism, genetics, number of pups and overall health, and can affect the results observed in many studies dealing with pollutants in marine mammals or biota. The comparisons, nonetheless, remain useful at the very least from the standpoint of examining trends.

Lee *et al.* (1996) studied male and female Steller sea lions from two locations. Alaskan blubber (n=29) and liver (n=28) samples were collected during 1976-1978 (Pitcher and Calkins, 1981). PCB and OCP data from this location were compared to data from Tatitlek (PWS) in the present study (Section 3.1 above). Lee *et al.* (1996) collected livers of 14 male Steller sea lion in 1981 from the Russian portion of the Bering Sea. Because no liver samples were collected at Olutorsky Gulf (R), liver data of Lee *et al.* (1996) were compared to liver data from St. Paul Island (BS).

Concentrations of ΣPCB in male Steller sea lion blubber and liver at the Alaska location (Lee *et al.*, 1996) were 17000 ng/g *lw* and 9000 ng/g *lw*, respectively (Table 3.11). Whereas the median concentrations in male animals from Tatitlek were 939 ng/g *lw* for blubber and 135 ng/g *lw* for liver. The temporal differences of ΣPCB concentrations of blubber and liver between samples from 1976-1978 and those of the present study are approximately one order of magnitude (Table 3.11). The highest median ΣPCB concentrations in male Steller sea lion blubber from St. Paul Island (BS) in the present study were about the same and up to 5x lower than those of the Alaskan location presented by Lee *et al.* (1996). Median ΣPCB concentrations in male liver tissues at the Russian location (4500 ng/g *lw*, Lee *et al.*, 1996) and St. Paul Island (BS) (591 ng/g *lw*, this study) differ by one order of magnitude. Median ΣPCB concentrations in female Steller sea lion reported by Lee *et al.* (1996) were 5300 and 4000 ng/g *lw* for blubber and liver (Table 3.11), respectively. In the present study ΣPCB concentrations in female tissues were 527 ng/g *lw* in blubber and 173 ng/g *lw* in liver tissues, or more than one order of magnitude lower.

Although many variables influence the data in both studies, a considerable and distinct decline in ΣPCB concentrations can be noted for the continental Alaskan and Russian Bering Sea regions over the last 20 years. PCB production and use were banned and restricted during the 1970s. Their worldwide use has declined steadily since that time. Animals in 1976-78 seemed to have been exposed on a greater scale to the then still prevalent PCBs in the environment and food chains. Long-range transport and distribution could have diluted and abiotic and biotic degradation and transformation could have lead to the decline of ΣPCB concentrations in Steller sea lions.

Lee *et al.* (1996) analyzed OCPs including p,p'-DDT, p,p'-DDD, p,p'-DDE, ΣDDTs, α-HCH, β-HCH, γ-HCH, ΣHCHs and HCB in blubber and liver tissues from the Alaskan location and in liver tissues from the Russian Bering sea. In the present study, OCPs were analyzed only in blubber tissues from each location. Therefore, a temporal comparison can only be made between the Alaskan locations, but not between the two Russian locations (Table 3.11).

Concentrations of p,p'-DDT, p,p'-DDD, p,p'-DDE and ΣDDTs in blubber of male and female Steller sea lion followed a similar declining trend as ΣPCBs for the Alaskan locations (Table 3.11). The decline of p,p-DDTs is usually directly connected to an increase of concentrations of its metabolites (or degradation products) which are more persistent. Lee *et al.* (1996) reported the highest concentrations for p,p'-DDE (5400 ng/g *lw*). In the present study, p,p'-DDE also had the highest concentration (709 ng/g *lw*) among p,p-DDT isomers (Table 3.11).

The use and production of DDTs have been restricted in the US since the 1970s and the overall use worldwide has noticeably decreased. A decrease in concentrations of DDTs over time has been shown in other environmental studies around the world, e.g., Helle *et al.* (1981) for ringed seals in the Baltic Sea, Muir *et al.* (1988) for ringed seals in the Northwest Atlantic Ocean, Tanabe *et al.* (1994) for Northern fur seals in the Northwest Pacific Ocean and Lieberg *et al.* (1995) for the California sea lion also in the Northwest Pacific Ocean.

Concentrations of α-HCH, γ-HCH and ΣHCHs showed a similar decline as PCBs between Alaskan locations in the present study and Lee *et al.* (1996). Concentrations of α-HCH, γ-HCH and ΣHCHs in male Steller sea lion blubber from Tatitlek were 47, 8 and 146 ng/g *lw*, respectively. Lee *et al.* (1996) reported concentrations of α -HCH, γ-HCH and ΣHCHs previously known as BHC at 300, 95 and 720 ng/g *lw*, respectively (Table 3.11). This steep decline can also be traced to the decreased worldwide use and implemented restriction of HCHs since the mid 1970s. The concentrations of β-HCH, the most stable HCH isomer, have also declined, but by only about 50% over the last 20 years for the Alaskan locations. Declines in HCH concentrations have been reported for pinnipeds (Tanabe *et al.*, 1994; Addison and Stobo, 2002) and cetaceans (Loganathan *et al.*, 1990; Muir *et al.*, 1996). However, β-HCH is known to be most persistent and highly accumulative in adipose tissue (ATSDR 2005). The data from the two studies suggest that the temporal decline of HCHs is not as pronounced as the decline of PCBs and DDTs. Technical HCH was banned from use in North America in the 1970s, but was still used in China until the 1980s and in India and the former Soviet Union until the 1990s as reported by Li *et al.* (1999). Compared to the other OCPs and PCBs the long-range transport potential and overall global distribution of α-HCH and γ-HCH is higher due to their low Koc and higher water solubility and volatility (Beyer *et al.*, 2000, Barber *et al.*, 2005). Relatively small temporal concentration differences of HCHs from 1976 to 2002 in the Alaskan region could potentially be explained by prolonged emissions from the Asian continent.

HCB concentrations were somewhat higher for male (3 ng/g *lw*) and female (7 ng/g *lw*) Steller sea lion blubber at the Alaskan location (Lee *et al.*, 1996), when compared to 2 and 3 ng/g *lw* for male and female animals from Tatitlek (present study). These differences in average HCB concentrations however seem rather small. If concentration ranges of HCB were compared between the two studies, animals studied by Lee *et al.* (1996) showed 2- to 3-fold higher concentrations than those in the present study (Table 3.11). A decrease in HCB concentrations was observed in Northern fur seals (Addison and Smith, 1998) and Beluga whales (Muir *et al.*, 1996).

Overall OCPs evaluated in this temporal comparison have shown a decrease in abundance in Steller sea lion blubber as have PCBs in blubber and liver tissues. A pronounced decline was noted for total PCBs, DDTs, α -HCH, γ-HCH and ΣHCHs. HCB and β-HCH concentrations also declined over the last 20 years, however, only to half to one third of the concentrations reported by Lee *et al.* (1996).

3.5. Acknowledgements

All Steller sea lion samples were obtained through the Tribal Ecosystem Office of St. Paul and Dr. Alan Springer, University of Alaska Fairbanks. Dr. D Wang assisted with analytical methods as well as sample preparation. This study was supported by Alaska SeaLife Center Steller Sea Lion Research Program with funds from National Marine Fisheries Services

Table 3.1. Steller sea lion (male and female) tissue and sample information

Sample ID	Location	Gender	Tissues		Date[a]
			Blubber	Liver	
Tatitlek # 4	Tatitlek (PWS)	♂	✓	✓	02/02/01
Tatitlek # 9	Tatitlek (PWS)	♂	✓	✗	02/02/01
Tatitlek # 3	Tatitlek (PWS)	♂	✓	✓	04/25/02
Tatitlek # 1	Tatitlek (PWS)	♂	✓	✓	04/07/02
Tatitlek # 10	Tatitlek (PWS)	♂	✓	✓	02/02/01
Tatitlek # 7	Tatitlek (PWS)	♂	✓	✓	02/02/01
Tatitlek # 8	Tatitlek (PWS)	♂	✓	✓	02/02/01
SSLSNP 2001-03	St. Paul (BS)	♂	✓	✓	09/16/01
SNPSLS9804	St. Paul (BS)	♂	✓	✓	10/21/98
SNPSLS9801	St. Paul (BS)	♂	✓	✓	09/22/98
SSLSNP 2001-05	St. Paul (BS)	♂	✓	✓	11/04/01
SNPSLS9802	St. Paul (BS)	♂	✗	✓	10/14/98
SNPSLS9803	St. Paul (BS)	♂	✓	✗	10/16/98
SNPSLS9902	St. Paul (BS)	♂	✓	✗	05/16/99
1-Ageev	Olutorsky Gulf (R)	♂	✓	✗	10/20/02
2-Ageev	Olutorsky Gulf (R)	♂	✓	✗	12/17/02
3-Ageev	Olutorsky Gulf (R)	♂	✓	✗	12/19/02
1-Testin	Olutorsky Gulf (R)	♂	✓	✗	10/25/02
2-Testin	Olutorsky Gulf (R)	♂	✓	✗	12/20/02
3-Testin	Olutorsky Gulf (R)	♂	✓	✗	12/20/02
1-Udalov	Olutorsky Gulf (R)	♂	✓	✗	12/01/02
2- Udalov	Olutorsky Gulf (R)	♂	✓	✗	12/06/02
3-Udalov	Olutorsky Gulf (R)	♂	✓	✗	no date
1-Vozikov	Olutorsky Gulf (R)	♂	✓	✗	no date
Tatitlek # 5	Tatitlek (PWS)	♀	✓	✓	02/02/01
Tatitlek # 6	Tatitlek (PWS)	♀	✓	✓	02/02/01
Tatitlek 3	Tatitlek (PWS)	♀	✓	✓	02/02/01
Tatitlek 1	Tatitlek (PWS)	♀	✓	✓	10/28/00
Tatitlek # 2	Tatitlek (PWS)	♀	✓	✓	04/07/02

PWS - Prince William Sound ; BS - Bering Sea ; R - Russia ; ✓- sample available ; ✗- no sample

[a] month-day-year

Table 3.2. Additional Steller sea lion (males) sample information

Sample ID	**Location**	**Tissues**		**Date**[b]
		Blubber	Liver	
SSL-2 (5)	Tatitlek (PWS)	✓	✓	10/28/2000
SSL-3	Tatitlek (PWS)	✓	✓	10/28/2000
SSL-4	Tatitlek (PWS)	✓	✓	10/28/2000
SNPSLS9802	St. Paul (BS)	✓	✓	10/14/1998
SNPSLS9901	St. Paul (BS)	✓	✓	4/29/1999
SNPSLS9808	St. Paul (BS)	✓	✓	Fall-98
SNPSLS2000-04	St. Paul (BS)	✓	✓	4/25/2000
Karin (SP-01-01-EJ)	St. Paul (BS)	✓	✓	3/23/2001
SP-01-00-EJ	St. Paul (BS)	✓	✖	5/31/2000

PWS - Prince William Sound ; BS - Bering Sea ; ✓- sample available ; ✖- no sample

[a] Data from chapter 2

[b] month-day-year

Table 3.3. Recoveries of 28 PCBs from lard (blubber surrogate)

PCB congeners	Recovery %	STD %	RSD %	MDL (3xStd) pg/g	Recovery range (%) by homology
PCB 8	77	27	35	84	
PCB 18	95	21	22	65	
PCB 28	117	24	21	76	
13C-PCB28	67	15	22	46	
PCB 52	109	16	15	50	67-131
PCB 44	127	15	12	48	
PCB 66	126	21	16	64	
PCB 81	128	23	18	72	
PCB 77	131	26	20	81	
PCB 101	116	13	11	40	
PCB 123	119	9	8	28	
13C-PCB123	76	4	6	14	
PCB 118	127	9	7	28	
PCB 114	118	5	4	16	
PCB 105	133	4	3	14	
PCB 153	120	11	9	34	
PCB 138	132	11	8	35	
PCB 126	127	11	8	33	76-133
PCB 128	120	11	9	33	
PCB 167	132	8	6	26	
PCB 156	111	4	3	11	
PCB 157	128	9	7	28	
PCB 169	125	5	4	17	
PCB 187	129	7	6	23	
PCB 180	130	6	5	19	
PCB 170	119	5	4	15	
PCB 189	128	6	5	19	
13C-PCB170	83	3	4	10	
PCB 195	126	10	8	30	
PCB 206	117	5	4	16	83-126
PCB 209	109	4	4	13	

STD - Standard deviation ; RSD - Relative standard deviation; MDL - Method detection limit

Table 3.4. Recoveries of 12 OCPs from lard (blubber surrogate)

OCP compounds	Recovery %	STD %	RSD %	MDL (3xStd) pg/g	Recovery range (%) by OCP group
α-BHC	121	6	5	19	
β-BHC	112	5	4	15	
γ-BHC	122	7	6	22	112-122
δ-BHC	119	9	8	29	
HCB	114	10	9	31	117
Heptachlor	79	16	20	50	79
o,p'-DDE	113	8	7	26	
p,p'-DDE	116	3	3	10	
o,p'-DDD	128	12	9	37	82-128
p,p'-DDD	100	5	5	16	
o,p'-DDT	84	10	12	32	
p,p'-DDT	82	10	12	31	

STD - Standard deviation ; RSD - Relative standard deviation; MDL - Method detection limit

Table 3.5. Recoveries of 28 PCBs from chicken liver (liver surrogate)

PCB congeners	Recovery %	STD %	RSD %	MDL (3xStd) pg/g	Recovery range (%) by homology
PCB 8	104	19	18	59	
PCB 18	107	14	13	43	
PCB 28	110	14	13	45	
13C-PCB28	126	9	7	27	96-130
PCB 52	120	25	21	78	
PCB 44	121	15	12	46	
PCB 66	96	7	8	23	
PCB 81	113	18	16	56	
PCB 77	130	15	12	49	
PCB 101	102	8	8	26	
PCB 123	97	14	15	45	
13C-PCB123	128	20	16	64	
PCB 118	109	7	7	22	
PCB 114	111	6	5	18	
PCB 105	98	5	5	15	
PCB 153	113	4	4	13	
PCB 138	113	6	5	19	96-128
PCB 126	120	7	6	22	
PCB 128	104	5	5	16	
PCB 167	112	5	4	15	
PCB 156	109	5	5	16	
PCB 157	110	6	5	17	
PCB 169	118	9	7	28	
PCB 187	96	10	10	31	
PCB 180	120	9	7	28	
PCB 170	103	7	7	21	
13C-PCB170	124	17	14	53	
PCB 189	119	6	5	18	
PCB 195	110	9	8	29	76-136
PCB 206	76	18	24	57	
PCB 209	136	29	22	92	

STD - Standard deviation ; RSD - Relative standard deviation; MDL - Method detection limit

Table 3.6. Sample information and total PCBs concentrations in the male/female Steller sea lion tissues

Gender	Location	Tissue		Lipids (%)	total PCBs concentrations (ng/g)			
					dw [a]		lw [b]	
					median (min-max)	iq R [c]		iq R [c]
Male	St. Paul (BS)	Blubber	n = 12	52	1145 (351-2994)	1164	2022 (777-3856)	1790
		Liver	n = 10	6	43 (16-51)	18	755 (198-1425)	446
	Olutorsky Gulf (R)	Blubber	n = 10	79	993 (694-2698)	389	1283 (847-3111)	263
	Tatitlek (PWS)	Blubber	n = 10	61	617 (258-1089)	388	956 ((374-1484))	596
		Liver	n = 11	6	24 (3-63)	26	238 (100- 973)	432
Female	Tatitlek (PWS)	Blubber	n = 5	74	305 (88-1833)	542	527 (103-2320)	720
		Liver	n = 5	7	12 (4-26)	12	173 (30-515)	212

[a] dw - dry weight ; [b] lw - lipid weight ; [c] iq R - interquartile range

Table 3.7. Persistent organic pollutants in Steller sea lion in the Northern Pacific Ocean area from 1992-2001

Species	Location	Tissue	Gender/Age (y)	total PCB (ng/g lw)	total DDTs (ng/g lw)	n	Publication
Steller sea lion	Prince William Sound / Bering sea	blubber	M, F / juvenile-adult	23000 ± 37000	20 000 ± 35 000 [a]	8	Varanasi et al. 1992
		liver	M, F / juvenile-adult	290 ± 340	210 ± 280 [a]	8	
Steller sea lion	Alaska	blubber	M / 0.9-12	17000 ± 12000	7 600 ± 5 400	12	Lee et al. 1996
			F / 2-25	5300 ± 5400	2 400 ± 2 300	17	
		liver	M / 2-12	9000 ± 6100	6 900 ± 6 600	13	
			F / 2-25	4000 ± 4000	2 500 ± 3 100	15	
	Bering Sea	liver	M / 2-16	4500 ± 1500	3 400 ± 1 600	14	
Steller sea lion	Southeast Alaska	blubber	M, F / juvenile	6600 ± 500 [b]	5 000 ± 600 [b]	3	Krahn et al. 1997 [c]
Steller sea lion	Prince William Sound	blubber	M, F / juvenile	1400 ± 700	1 300 ± 600	19	Krahn et al. 2001
	Southeast Alaska			1600 ± 600	1 400 ± 1 100	10	

[a] DDTs as combination of o,p-DDT, o,p-DDE, o,p-DDD, p,p-DDT, p,p-DDE, and p,p-DDD

[b] Concentrations reported in μg/g wet weight

[c] Krahn et al. 1997, "from Steller Sea Lion Recovery Investigations in Alaska 1995–1996"

Table 3.8. Median concentrations of single PCB congeners and total PCBs (ng/g *lw*) in male and female Steller sea lion

			Lipid content	PCB Congeners											
Location	**Gender**	**n**	**%**	**8**	**18**	**28**	**52**	**44**	**66**	**77**	**101**	**123**	**118**	**114**	**105**
Olutorsky Gulf (R)	♂	10	79	1	7	20	77	21	43	1	88	138	164	4	48
Tatitlek (PWS)		7	79	3	7	20	66	11	34	0[a]	51	0[a]	118	4	25
St. Paul (BS)		6	70	3	6	25	68	12	120	0[a]	37	0[a]	465	9	166
Tatitlek (PWS)	♀	5	74	4	6	18	40	13	31	0[a]	41	0[a]	60	2	15

PWS - Prince William Sound ; BS - Bering Sea ; R - Russia

[a] Data as median 0 values

			Lipid content	PCB Congeners												median PCBs
Location	**Gender**	**n**	**%**	**153**	**138**	**128**	**167**	**156**	**157**	**187**	**180**	**170**	**195**	**206**	**209**	**(ng/g lw)**
Olutorsky Gulf (R)	♂	10	79	324	207	35	2	2	3	21	78	21	2	2	1	1257
Tatitlek (PWS)		7	79	277	158	26	2	1	1	30	59	17	1	2	0[a]	939
St. Paul (BS)		6	70	1052	568	140	10	10	2	34	213	72	5	5	2	3173
Tatitlek (PWS)	♀	5	74	110	62	8	1	0[a]	1	9	19	5	1	1	1	527

PWS - Prince William Sound ; BS - Bering Sea ; R - Russia

[a] Data as median 0 values

Table 3.9. Median OCP compound concentrations ng/g *lw* in Steller sea lion (male and female) blubber tissues

Location	Gender		Lipids (%)	OCP concentrations (ng/g lw)									
				ΣHCHs (range)	iq R [a]	HCB (range)	iq R [a]	Heptachlor (range)	iq R [a]	Σ o,p DDTs (range)	iq R [a]	Σ p,p DDTs (range)	iq R [a]
Tatitlek $_{(PWS)}$	Male	n = 7	81	146 (76-235)	77	2 (1-4)	1	19 (0-31)	22	39 (10-80)	32	952 (406-2028)	951
St. Paul $_{(BS)}$		n = 6	60	424 (277-639)	322	4 (2-6)	3	21 (10-29)	7	28 (12-126)	80	3023 (1091-6322)	1163
Olutorsky Gulf $_{(R)}$		n = 10	79	227 (146-504)	91	1 (2-4)	1	9 (6-12)	1	79 (17-153)	50	1448 (501-2387)	704
Tatitlek $_{(PWS)}$	Female	n = 5	74	56 (19-621)	81	3 (2-5)	1	14 (5-28)	9	9 (3-29)	6	489 (54-1600)	719

[a] iq R - interquartile range

Table 3.10. Average toxic equivalents (TEQs) for male/ female Steller sea lion from Tatitlek (PWS), St. Paul Island (BS) and Olutorsky Gulf (R)

PCBs	TEFs	Tatilek (male)	St. Paul Island	Olutorsky Gulf	Tatilek (female)	Tatilek (male)	St. Paul Island	Tatilek (female)
		Blubber tissues				Liver tissues		
77	0.0001	0.01	0.1	2.2	1.0	1.6	nd	nd
81	0.0001	0.01	nd	nd	nd	nd	nd	nd
105	0.0001	2.9	15.3	5.1	3.5	37.7	3.8	1.1
114	0.0005	1.7	5.0	2.1	1.3	0.6	0.9	0.3
118	0.0001	10.8	46.6	17.1	9.9	0.04	7.0	3.1
123	0.0001	nd	2.0	12.2	1.0	nd	nd	nd
126	0.1	nd	nd	nd	nd	0.38	nd	nd
156	0.0005	0.8	5.9	1.3	0.9	nd	1.2	0.01
157	0.0005	0.9	2.9	1.4	0.8	nd	nd	nd
167	0.00001	0.02	0.1	0.02	0.02	nd	nd	nd
Sum TEQs		17	78	41	18	40	13	5

nd - not detected

Table 3.11. Concentrations of total PCBs, DDTs (p,p-DDD, p,p-DDE, p,p-DDT), HCHs

Location	Gender	Tissue	n	Lipids (%)	total PCBs [a] (ng/g lw)	p,p'-DDD [a]	p,p'-DDE [a]	p,p'-DDT [a]	ΣDDTs [a]	Reference
Alaska	M	blubber	12	74	17 000 (5 700-4 100)	960 (410-2 200)	5 400 (1 500-13 000)	1 200 (410-2 500)	7 600 (2 800-17 000)	Lee et al. 1996
	F		17	82	5 300 (570-16 000)	300 (22-710)	1 800 (87-5 100)	350 (82-980)	2 400 (190-6 500)	
	M	liver	13	6	9 000 (3 200-25 000)	910 (170-2 900)	6 000 (1 800-22 000	19 (nd-130)	6 900 2 100-25 000)	
	F		15	6	4 000 (450-13 000)	360 (63-1 000)	2 100 (31-8 600)	24 (nd-280)	2 500 (120-9 600)	
Russian Bering Sea	M	liver	14	4	4 500 (3 200-8 500)	430 (280-750)	2 900 (1 300-6 700)	28 (nd-52)	3 400 (1 600-7 500)	

Location	Gender	Tissue	n	Lipids (%)	total PCBs [b] (ng/g lw)	p,p'-DDD [b]	p,p'-DDE [b]	p,p'-DDT [b]	ΣDDTs [b]	
Tatitlek (PWS)	M	blubber	7	81	939 (374-1432) [b]	116 (62-158)	709 (305-1754)	50 (25-114)	952 (406-2028)	Present study
	F	blubber	5	74	527 (103-2320)	48 (10-349)	376 (270-1391)	56 (4-100)	489 (54-1600)	
	M	liver	6	7	135 (100- 238)	\	\	\	\	
	F	liver	5	7	173 (30-515)	\	\	\	\	
St. Paul Island (BS)	M	blubber	6	60	3173 (1814-3856)	254 (175-407)	2382 (854-5366)	217 (50-741)	3023 (1091-6322)	
		liver	10	6	591 (198-1000)	\	\	\	\	
Olutorsky Gulf (R)	M	blubber	10	79	1257 (847-3111)	168 (86-451)	1080 (371-1652)	61 (28-131)	1448 (501-2387)	

lw - lipid weight ; a - average (min-max) ; b - median (min-max) ; nd - less than detection limit

Location	Gender	Tissue	n	Lipids (%)	α-BHC [a]	β-BHC [a]	γ-BHC [a]	ΣBHCs [a]	HCB [a]	Reference
Alaska	M	blubber	12	74	300 (140-550)	320 (150-530)	95 (30-230)	720 (380-1 200)	3 (nd-8)	Lee et al. 1996
	F		17	82	220 (130-310)	150 (30-440)	51 (26-130)	410 (190-710)	7 (2-17)	
	M	liver	13	6	170 (75-360)	250 (45-700)	45 (5-85)	460 2(80-950)	3 (nd-7)	
	F		15	6	130 (55-190)	160 (15-410)	30 (10-65)	310 (85-600)	3 (nd-7)	
Russian Bering Sea	M	liver	14	4	110 (73-210)	170 (75-420)	34 (21-72)	310 (210-590)	7 (2-21)	

Location	Gender	Tissue	n	Lipids (%)	α-HCH [b]	β-HCH [b]	γ-HCH [a]	ΣHCHs [b]	HCB [b]	
Tatitlek (PWS)	M	blubber	7	81	47 (5-70)	103 (48-148)	8 (3-23)	146 (76-235)	2 (1.3-4)	Present study
	F	blubber	5	74	10 (9-34)	44 (8-570)	3 (1-14)	56 (19-621)	3 (1.8-4.7)	
	M	liver	6	7	\	\	\	\	\	
	F	liver	5	7	\	\	\	\	\	
St. Paul Island (BS)	M	blubber	6	60	22 (14-58)	297 (206-595)	10 (3-14)	424 (277-639)	4 (1.7-5.6)	
		liver	10	6	\	\	\	\	\	
Olutorsky Gulf (R)	M	blubber	10	79	36 (21-50)	168 (99-228)	16 (7-24)	227 (146-504)	2 (1.6-4.4)	

lw - lipid weight ; a - average (min-max) ; b - median (min-max) ; nd - less than detection limit

HCH - Hexachlorocyclohexane which was formally known as benzene hexachloride (BHC).

Figure 3.1. A map of the Steller sea lion habitat and sample origin in the northern Pacific Ocean

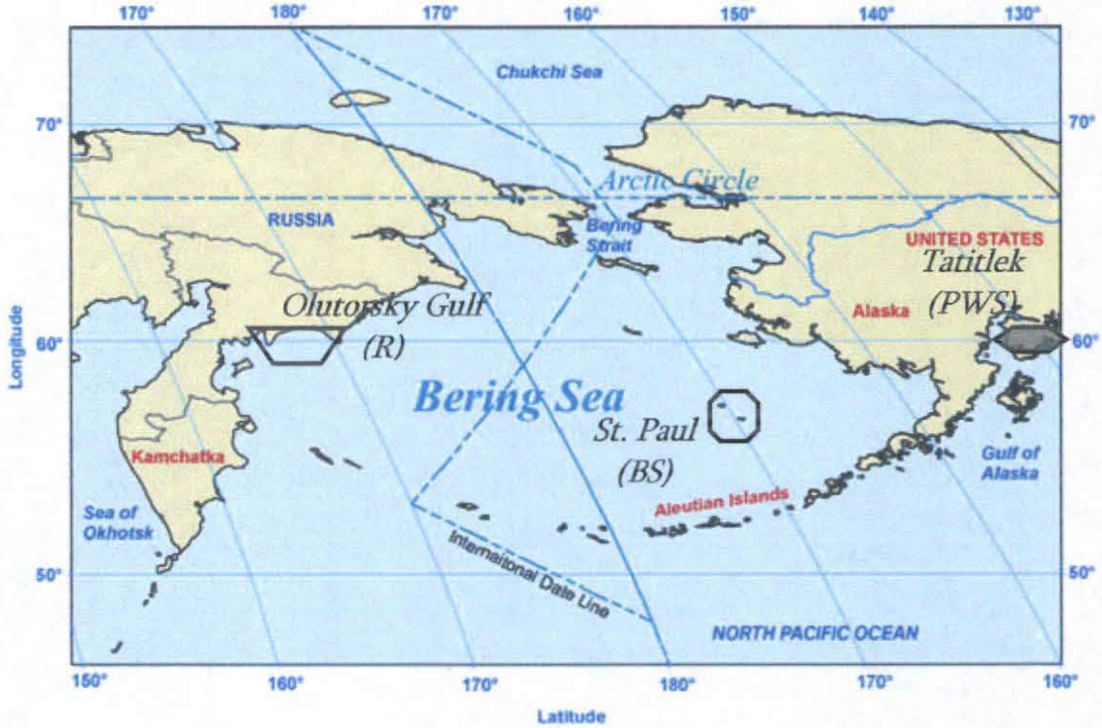


Figure 3.2. Median concentrations (µg/g *lw*) of total PCBs in Steller sea lions from Tatitlek (PWS), St Paul Island (BS) and Olutorky Gulf (R)

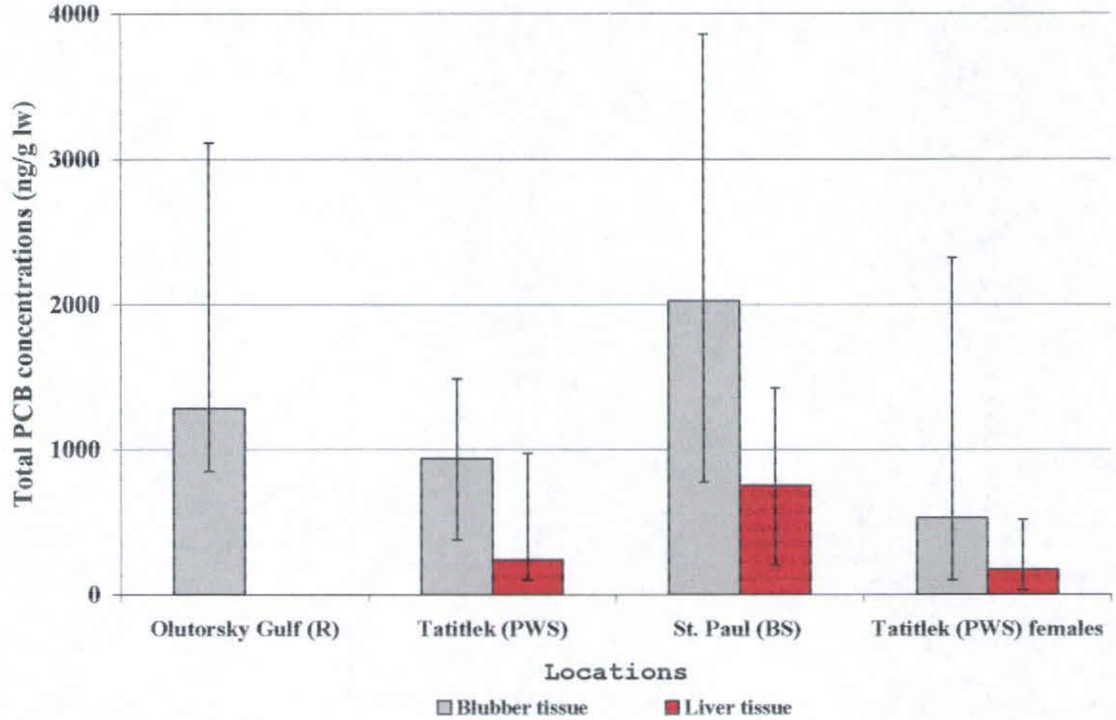


Figure 3.3. Box-plot of median, quartiles and single data points PCB concentrations (μg/g *lw*) for male/female Steller sea lion tissues from two locations

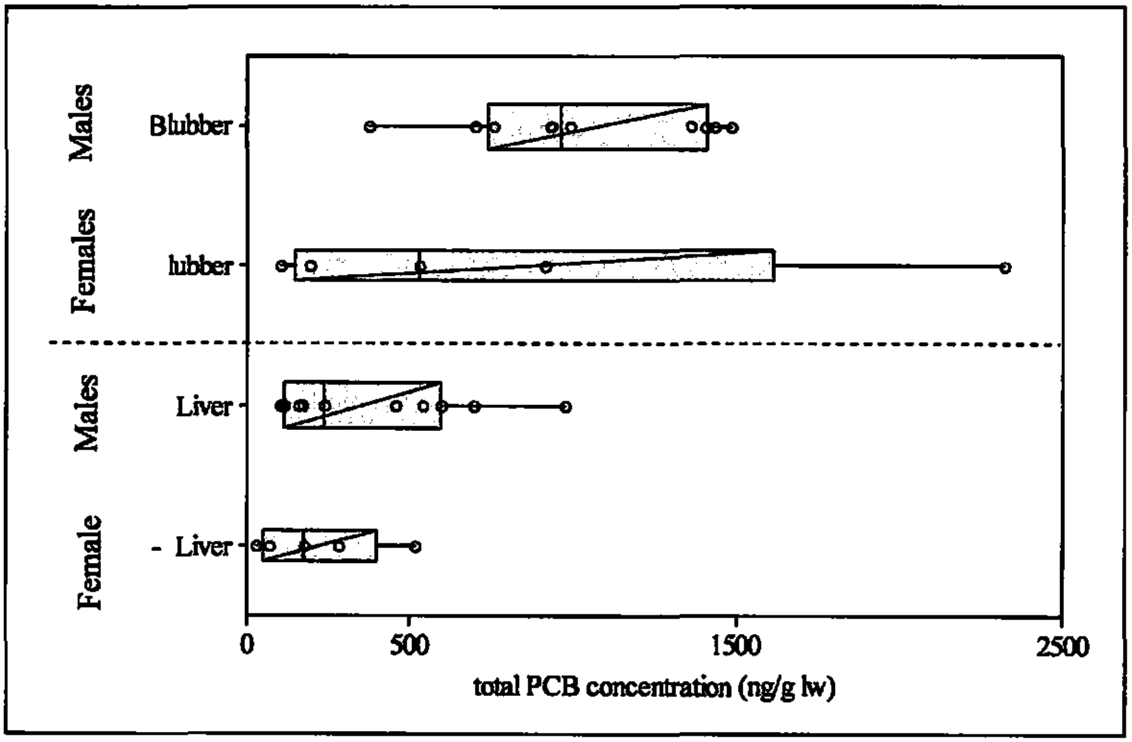


Figure 3.4. Profiles (median) of 28 PCB congeners in Steller sea lion blubber and liver from all locations (male and female)

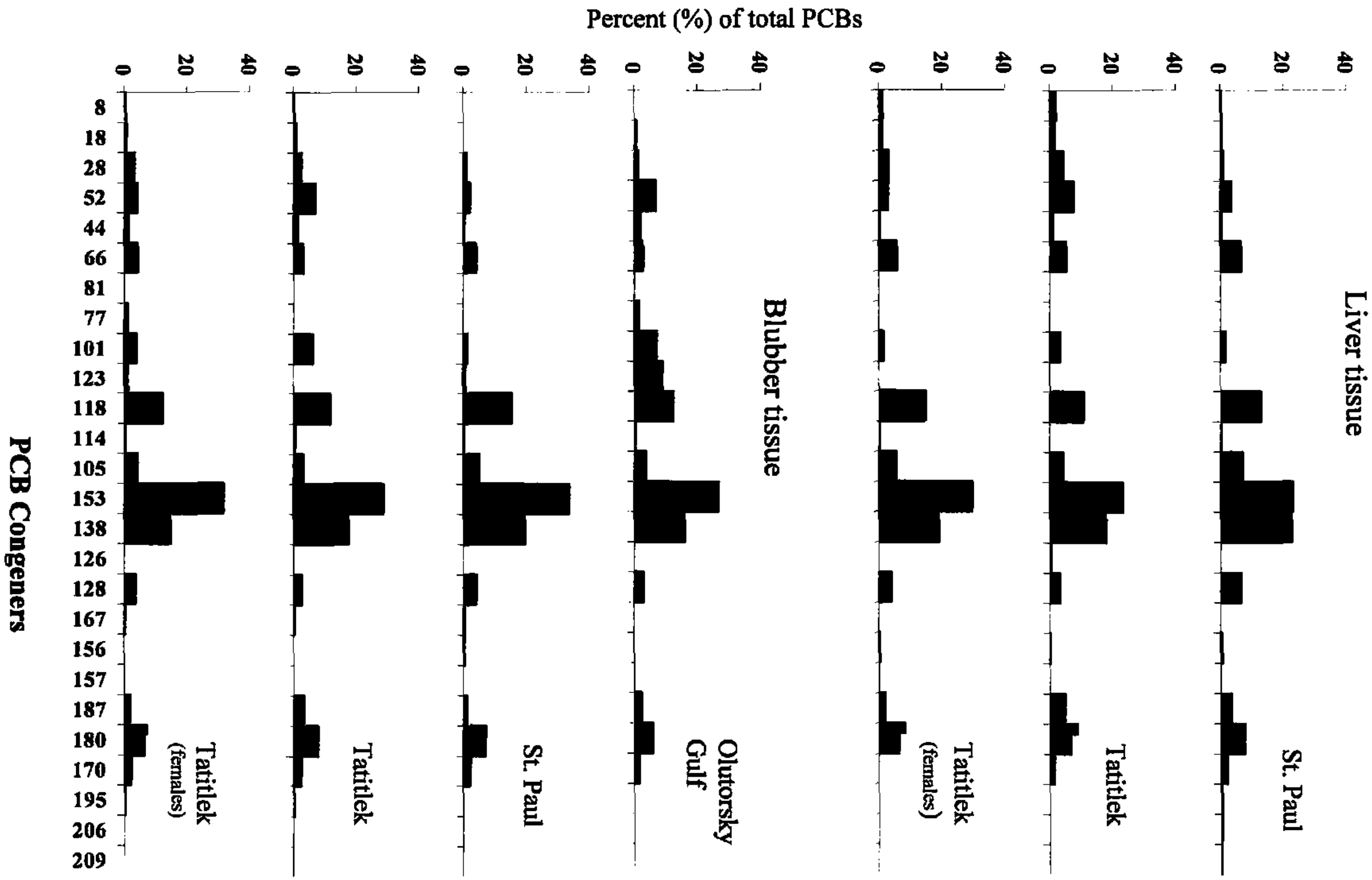


Figure 3.5. Aroclor 1016, 1242, 1248a, 1248b, 1260, 1254a and 1254b as well as Steller sea lion tissue homologues for 28 selected PCB congeners

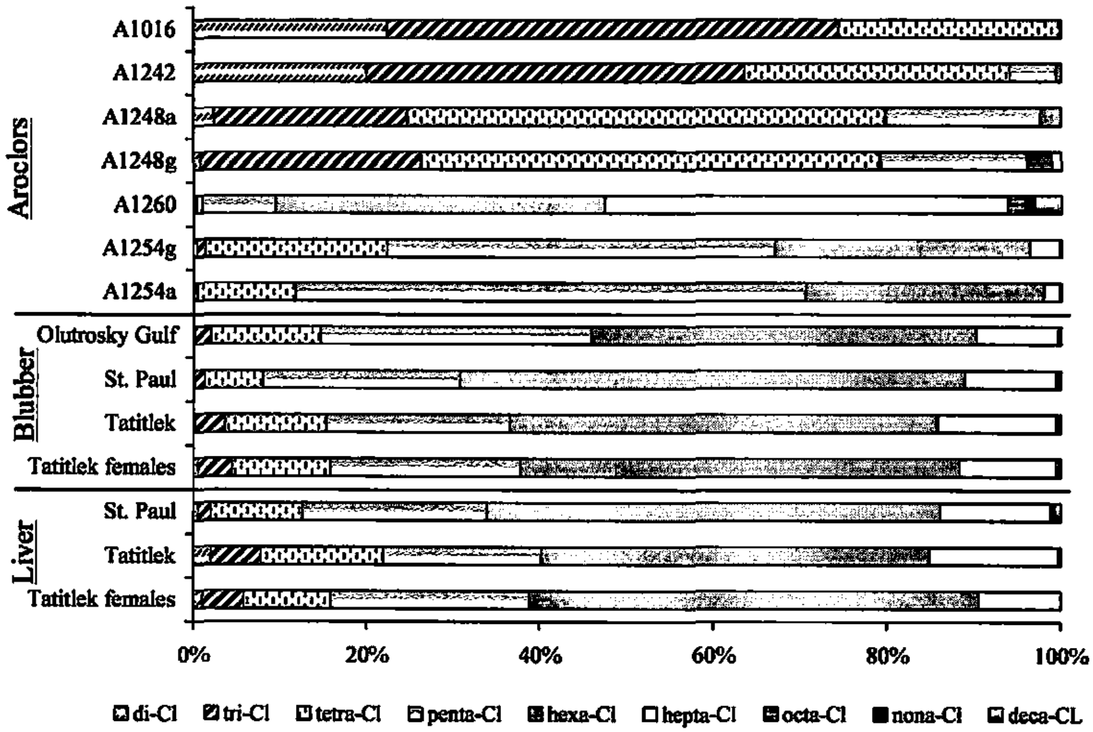
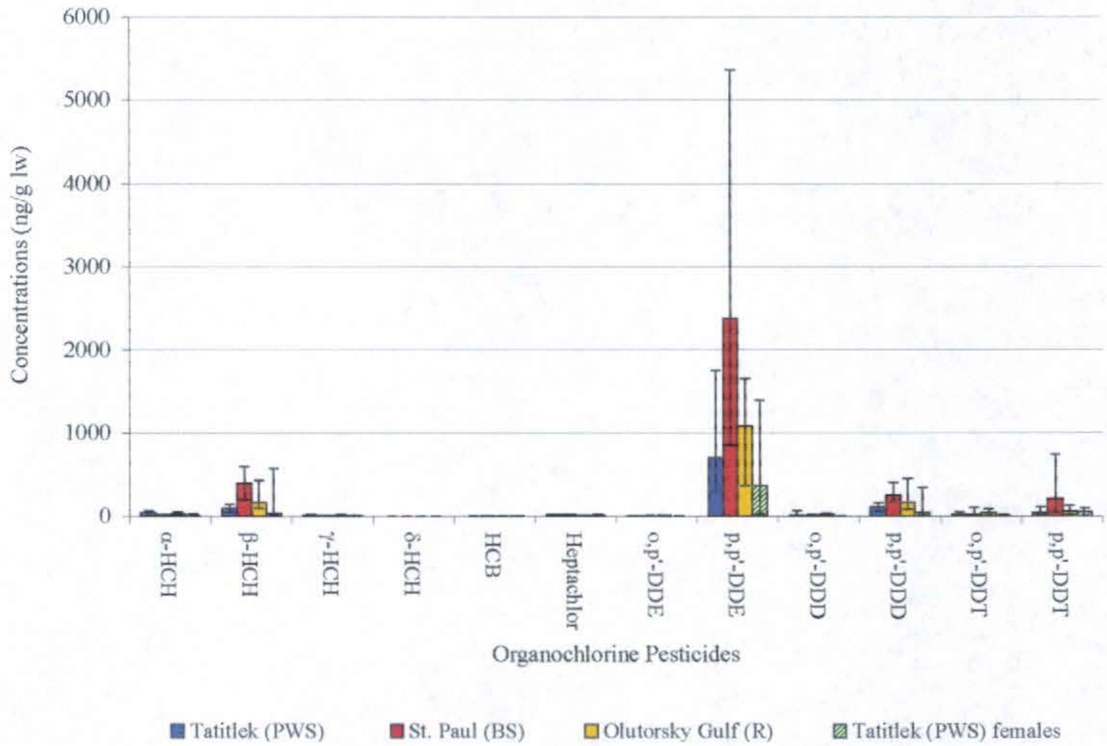


Figure 3.6. Location comparison of individual OCP concentrations for Tatitlek (PWS), St Paul Island (BS) and Olutorsky Gulf (R)



CHAPTER 4

CONCLUSIONS AND FUTURE OUTLOOK

4.1. Conclusions

Extensive literature review suggests that only limited data exists for POPs in Steller sea lions. It is suspected that POPs pollution is a cause of the Steller sea lion population decline as well as being responsible for a slowed population recovery. Therefore, the goal of my thesis research was to quire new data on POPs contamination in Steller sea lions, thus improving on understanding of the current condition of this species and their habitat. Because they are listed as an endangered species the ultimate goal is to provide information that will aid in the recovery of Steller sea lions.

POPs were detected in all of the analyzed samples described in chapters 2 and 3. Chapter 2 reported data for blubber, liver and kidney samples from males Steller sea lions and placenta samples from female Steller sea lions. A method was established for 145 individual PCB congeners of which 57 were detected in blubber, 65 in liver samples, 38 in kidney and 36 in placenta tissues. Samples were collected from two locations: Tatitlek (Prince William Sound, Alaska) and St. Paul Island (Pribilof Islands, Bering Sea). No significant differences in concentrations of PCBs were found between these two locations. The average concentrations of PCBs in blubber ($n = 9$) and liver ($n = 10$) were below the immunotoxic threshold of 17 μg/g *lw* for blubber and physiological toxic threshold of 6.6-11 μg/g *lw* for liver. Two males [SSL-2 (5) and SSL9802SNP], however, showed concentrations of ΣPCBs of 8.0 μg/g and 6.6 μg/g *lw*, respectively, thus within the physiological toxic threshold of liver tissues. Four dioxin-like PCB congeners

(81, 105, 118, and 157) were detected in the Steller sea lion tissues analyzed. The calculated ΣPCB-TEQs for each of the four tissue types as well as the averages were below the theoretical threshold for toxic effects (520 pg/g *lw*).

Chapter 3 presents data derived from blubber and liver samples of male and female Steller sea lions. They were analyzed for 28 PCB congeners as recommended by the USEPA, WHO and NOAA. The tissues were also analyzed for 12 OCPs: o,p'-DDT, p,p'-DDT, o,p'-DDD, p,p'-DDD, o,p'- DDE, p,p'-DDE, HCH isomers (α-, β-, γ- and δ-HCH), heptachlor and HCB. Steller sea lion tissues analyzed in this part of the study were collected at three distinct locations, Tatitlek (Prince William Sounds, Alaska), St. Paul Island (Pribilof Islands, Bering Sea) and Olutorsky Gulf (Russia). The overall POP concentrations found in the present study were highest at St. Paul Island followed by Olutorsky Gulf and lowest at Tatitlek. Significantly higher concentrations of ΣPCBs were found in males from St. Paul and Olutorsky Gulf than those from Tatitlek. Significantly higher β-HCH, ΣHCHs, HCB, heptachlor, o,p'-DDE, p,p'-DDE, p,p'-DDT, p,p'-DDD, and Σp,p-DDTs were found in samples from St. Paul than Tatitlek or Olutorsky Gulf. A gender comparison of blubber and liver tissues of animals from Tatitlek (males, n = 7; females n = 5) showed significantly higher concentrations for o,p'-DDD, o,p'- DDT and Σo,p'-DDTs in males than in females. Concentrations of p,p'-DDD, p,p'- DDE, p,p'- DDT and Σp,p'-DDTs were also higher in males at Tatitlek, but the differences were not statistically significant.

All ΣPCB concentrations in animal blubber and liver tissues reported here were below both the physiological toxic threshold of 6.6-11 μg/g *lw* of liver and the immunotoxic threshold of 17 μg/g for blubber (Kannan *et al.*, 2000). The calculated

ΣPCBs-TEQs for both male and female Steller sea lion blubber and liver tissues did not reach the toxic effect threshold level of 520 pg/g *lw* (Kannan *et al.,* 2000)

The temporal comparison between concentrations of different POPs (PCBs, HCHs, DDTs and HCB) in blubber and liver samples observed in this study with those from 1976-1981 (Lee *et al.*, 1996) showed an overall trend of sharply decreasing concentrations (about one order of magnitude) over the last 20 years. Only HCB and the β-HCH isomer concentrations showed somewhat lesser declines, to a half or 1/3 the concentrations reported by Lee *et al.* (1996).

Animals in this study overall have low ΣPCB-TEQs except for two animals which showed concentrations within the physiological toxic threshold of 6.6-11 μg/g *lw* in the liver of marine mammals (Kannan *et al.,* 2000). These findings suggest that POP exposure and bioaccumulation might represent a minor contribution to the decline of the Steller sea lion western stock. It has to be noted that the 29 PCB congers and 12 OCP compounds evaluated in this study showed a substantial temporal decline over a period of ~20 years and may have had toxocological effects during the time of peak emissions. Furthermore the impact of new (not yet detectable) POPs and the 'synergistic toxicity' of POP mixtures could result in significant adverse effects on the Steller sea lion population.

Results from this study provided new data for POPs in Steller sea lions and suggest an overall decline in the concentrations of the POPs discussed in this thesis in regions of the Northern Pacific Ocean. These results are cosistent with other temporal studies on marine mammals worldwide (Helle *et al*., 1981; Muir *et al*., 1988; Loganathan *et al.*, 1990; Tanabe *et al*., 1994; Muir *et al.*, 1996; Lieberg *et al*., 1995; Addison and Stobo, 2002). The decline in POPs concentrations is attributable to the restriction and ban of many of these substances that entailed the large scale emissions of these exclusively man-made chemicals. Their physical dispersion, chemical (abiotic) degradation as well as biotic transformations have enhanced the environmental recovery from their original concentrations. Without the restriction and ban of POPs a decline in this abundance would have been unlikely and an increase in their adverse environmental impact would be the most likely scenario.

4.2. Future Outlook

POPs analyzed and discussed in this study have now been restricted or banned in many countries. These efforts continue, e.g., for 12 of the most known and recalcitrant POPs, the so called "dirty dozen", through the Stockholm Convention on Persistent Organic Pollutants (2001) administerd under the auspices of the United Nations Environment Programme (UNEP). All POPs mentioned in this thesis, but especially PCBs and DDTs, have been studied extensively since the discovery of their accumulation potential in the 1960s. Unfortunately, their fate and distribution in the environment remain unknown to a great extent. Further studies and well planned monitoring projects could help uncover the long-term trends, environmental fate and bioavailability of PCBs and DDTs for water, soil, sediment and biota. In vitro studies could also enhance on the knowledge of toxicity and adverse effects of PCBs, DDTs and other POPs.

A multitude of "newer" POPs such as tetrabromobisphenol-A (TBBPA), perfluorooctane sulfonate (PFOS); chlorinated naphthalenes (PCNs), short-chain chlorinated paraffins and polybrominated biphenyl ethers (PBDEs), have also been used or continue to be produced and used. Several of these "newer" POPs show high accumulation potential and have been identified in Arctic regions. PBDEs, for example, have been used in similar applications as PCBs and in fact replaced PCBs after their ban in 1976. More and more evidence is found that PBDEs show similar trends and fate in the environment. PBDEs, however, adsorb less strongly to solid particles and their bioaccumulation and magnification potential is lower. Bromine, has a larger atomic radius and lower electronegativity than chlorine, making it more polar (Morris and Boyd, 1992). PCBs and DDTs (and other known persistent organic pollutants) could act as model compounds for similar structural pollutants (chemical structures, water solubility, K_{oc}, K_{ow}, etc.) in the assessments of their potential impacts. For example, California passed a state-wide ban on flame retardant PBDEs. Two of the main commercial grades of PBDEs (penta and octa) were banned from use in 2006 in California and Europe. Decabrominated diphenyl ether (decaBDE) was exempt from this ban and continues to be produced and used. It is thought that due to its high molecular weight decaBDE may not enter living cells (Hardy *et al.*, 2000). However, it is unclear to what extent decaBDE is degraded by microorganisms in sediments and soil and transformed into lower mass brominated BDE congeners. Further restrictions for POPs as well as replacement with less toxic and accumulative compounds are needed. In the meantime, continued monitoring of POPs in marine mammals is needed to assist Steller sea lion population management and recovery.

Not only are "newer" POPs of interest, but also the risk assessment of mixtures of various chemicals found in aquatic organisms (Walter *et al.*, 2002; Brack, 2003; McCarth and Borgert, 2006). Today approximately 80,000 man-made chemicals are used in a variety of agricultural and industrial applications and 2-3 new chemicals enter the market every day.

Marine mammals such as the Steller sea lion are known to be hunted by aboriginal communities in the Artic, to enrich their diet (subsistence hunting and traditional uses of wild foods) (Hansen, 2000). In fact Hansen, (2000) proposed that consumption of marine mammals (and other wildlife) is the major source of dietary environmental contaminants in the Artic. Artic population groups are therefore more likely to be exposed to environmental contaminants on a much higher level though their diet than most other human populations worldwide. Ultimately, POPs enter the human diet and long-term adverse human health effects have been shown in connection to POP exposure (Miller *et al.,* 2004; Muir *et al.,* 2005). Increasing evidence of adverse health trends is also being uncovered in human reproduction related tissues, e.g., testicular cancer and female breast cancer (Bonefeld-Jørgensen, 2004). Thus, data collection and evaluation of Steller sea lions used as a food source by aboriginal people may aid evaluations of human health conditions and advance future management of POPs.

REFERENCES

Abramowicz DA. Aerobic and anaerobic degradation of PCBs: A review. Crit. Rev. Biotchnol. 1990; 10: 241-248

Addison RF, Smith TG. Trends in organochlorine residue concentrations in ringed seal (*Phoca hispida*) from Holman, northwest Territories, 1972-1991, Arctic 1998; 51: 253-261

Addison RF, Stobo WT. Trends in organochlorine residue concentrations and burdens in grey seals (*Halichoerus grypus*) from Sable IS., NS, Canada, between 1974-1994. Environmental Pollution 2001; 112: 505-513

Agency for Toxic Substances and Disease Registry (ATSDR). Toxicological profile for chlorinated dibenzo-p-dioxins (CDDs). Atlanta, GA: U.S. Department of Health and Human Services, Public Health Service 1998

Agency for Toxic Substances and Disease Registry (ATSDR). Toxicological profile for polychlorinated biphenyls (PCBs). Atlanta, GA: U.S. Department of Health and Human Services, Public Health Service 2000

Agency for Toxic Substances and Disease Registry (ATSDR). Toxicological profile for DDT, DDE, DDD. Atlanta, GA: U.S. Department of Health and Human Services, Public Health Service 2002

Agency for Toxic Substances and Disease Registry (ATSDR). Toxicological profile for Hexachlorobenzene. Atlanta, GA: U.S. Department of Health and Human Services, Public Health Service 2002

Agency for Toxic Substances and Disease Registry (ATSDR). Toxicological profile Heptachlor/Heptachlor epoxide. Atlanta, GA: U.S. Department of Health and Human Services, Public Health Service 2005

Aguilar A, Borrell A. Age- and sex-related changes in organochlorine compound concentrations in fin whales (*Balaenoptera physalus*) from the eastern North Atlantic. Mar. Environ. Res. 1988; 25: 195-211

Aguilar A, Borell A, Reijnders PJH. Geographical and temporal variation in levels of organochlorine contaminants in marine mammals. Mar. Environ. Res. 2002; 53: 425-452

Aguilar A, Borrell A. Heterogeneous distribution of organochlorine contaminants in the blubber of baleen whales: Implications for sampling procedures. Mar. Environ. Res. 1991; 31: 275-286

AMAP, 2003. AMAP Assessment 2002: The influence of global change on contaminant pathways to, within, and from the Arctic. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. xi+65 pp.

Bailey RE. Global hexachlorobenzene emissions. Chemosphere 2001; 43: 167-182

Ballschmitter K, Zell M. (1980). Analysis of polychlorinated biphenyls (PCB) by glass capillary gas chromatography. Composition of technical Aroclor- and Clophen-PCB mixtures. Fresenius J. Anal. Chem. 302, 20-31

Barber JL, Sweetman AJ, Van Wijk D, Jones KC. Hexachlorobenzene in the global environment: Emissions, levels, distribution, trends and processes. Sci. Total. Environ. 2005; 349, 1-44

Barron MG. Bioaccumulation and bioconcentration in aquatic organisms. In D.J.

Hoffman, B.A. Rattner, G.A. Burton, Jr. and J. Cairns, Jr. [eds.], Handbook of Ecotoxicology, pp. 652-666. Lewis Publisher, Boca Ranton, Florida 1995

Barron MG, Galbraith H, Beltman D. Comparative reproductive and developmental toxicology of PCBs in birds. Comp. Biochem. Physiol., Part C: 1995; 112(1): 1-14

Becker PR. Concentration of chlorinated hydrocarbons and heavy metals in Alaska Arctic marine mammals. Mar. Pollut. Bull. 2000; 40(10) 819-829

Beckmen KB, Blake JE, Ylitalo GM, Stott JL, O'Hara Todd. Organochlorine contaminant exposure and associations with hematological and humoral immune functional assays with dam age as a factor in free-ranging northern fur seal pups (*Callorhinus ursinus*). Mar. Pollut. Bull. 2003; 46(5): 594–606

Ben-Dyke R, Sanderson D, Noakes D. Acute toxicity data for pesticides (1970). World Rev. Pestic. Cont. 1970; 9: 119-127

Bergman A, Olsson M, Reiland S. Skull-bone lesions in the Baltic grey seal (*Halichoerus grypus*). Ambio 1992; 21: 517-519

Berggrena P, Ishaq R, ZebUhr Y, Naf C, Bandh C, Broman D. Patterns and levels of organochlorines (DDTs, PCBs, non-*ortho* PCBs and PCDD/Fs) in male harbour porpoises (*Phocoena phocoena*) from the Baltic Sea, the Kattegat-Skagerrak Seas and the West Coast of Norway. Mar. Pollut. Bull. 1999; 38: 1070–1084

Berman E, Schlicht M, Moser VC. A multidisciplinary approach to toxicological screening: I. Systemic toxicity. J. Toxicol. Environ. Health. 1995; 45: 127-143

Bernt KE, Hammill MO, Lebeuf M, Kovacs KM. Level and patterns of PCBs and OC pesticides in harbour and grey seals from the St Lawrence Estuary, Canada. Sci.

Tot. Env. 1999; 244: 243-262

Beyer A, Mackay D, Matthies M, Wania F, Webster E. Assessing long-range transport potential of persistent organic pollutants. Environ. Sci. Technol. 2000; 34: 699-703

Bickham JW, Patton JC, Loughlin TR. High variability for control-region sequences in a marine mammal: Implications for conservation and biogeography of Steller sea lions (*Eumetopias jubatus*). J.Mammal. 1996; 77(1): 95-108

Bjerregaard P, Hansen JC. Organochlorines and heavy metals in pregnant women from the Disko Bay area in Greenland. Sci. Total. Environ. 2000; 245: 195-202

Bonefeld-Jørgensen EC. The human health effect programme in Greenland, a review. Sci. Total Environ. 2004; 331: 215–231

Boon JP, Reijnders PJH, Dols J, Wensvoort P, Hillebrand MTJ. The kinetics of individual polychlorinated biphenyl congeners in female harbour seals (*Phoca vitulina*), with evidence for structure-related metabolism. Aquat. Toxicol. 1987; 10: 307-324

Boon JP, van der Meer J, Allchin CR, RJ Law, Kungsøyr J, Leonards PEG, Spliid H, Storr-Hansen E, Mckenzie C, Wells DE. Concentration-dependent changes of PCB patterns in fish-eating mammals: Structural evidence for induction of cytochrom P450. Arch. Environ. Contam. Toxicol. 1997; 33: 298-311

Borgå K, Gabrielsen GW, Skaare JU. Biomagnification of organochlorines along a Barents Sea food chain. Envrion. Poll. 2001; 113: 187-198

Brack W. Effect-directed analysis: a promising tool for the identification of organic toxicants in complex mixtures? Anal. Bioanal. Chem. 2003; 377: 397–407

Braham, H.W., R.D. Everitt, and D.J. Rugh. 1980. Northern sea lion population decline in the eastern Aleutian Islands. J. Wildl. Manage. 1980; 44: 25-33

Bristol DW, Crist HL, Lewis RG. Chemical analysis of human blood for assessment of environmental exposure to semivolatile organochlorine chemical contaminants. J. Anal. Toxicol. 1982; 6: 1-7

Brown LM, Pottern LM, Hoover RN, Devesa SS, Aselton P, Flannery JT. Testicular cancer in the United States: trends in incidence and mortality. Int. J. Epidemiol. 1986; 15: 164-170

Brown Jr. JF, Bedard DL, Brennan MJ, Carnahan JC, Feng H, Wagner RE. Polychlorinated biphenyl dechlorination in aquatic sediment. Science 1987A; 236: 709-712

Brown Jr. JF, Wagner RE, Feng H, Bedard DL, Brennan MJ, Carnahan JC, May RJ. Environmental dechlorination of PCBs. Environ. Toxicol. Chem. 1987B; 6: 579-593

Bruckner JV, Khanna KL, Cornish HH. Biological responses of the rat to polychlorinated biphenyls. Toxicol. Appl. Pharmacol. 1973; 24: 434-448

Bruhn R, Kannan N, Petrick G, Schulz-Bull DE Duinker, JC. CB Pattern in the harbour porpoise: Bioaccumulation, metabolism and evidence for cytochrome P450 IIB activity. Chemosphere 1995; 31(7): 3721-3732

Burse VW, Korver MP, Needham LL. Gas chromatographic determination of polychlorinated biphenyls (as Aroclor 1254) in serum: Collaborative study. J. Assoc. Off. Anal. Chem. 1989; 72(4): 649-659

Burse VW, Head SL, Korver MP. Determination of selected organochloride pesticides and polychlorinated biphenyls in human serum. J. Anal. Toxicol. 1990; 14: 137-142

Calkins DG, Pitcher KW. Population assessment, ecology and trophic relationships of Steller sea lions in the Gulf of Alaska. Final Report Outer Continental Shelf Environmental Assessment Program. 1982; 129pp.

Cameron G, Burgess F. The toxicity of 2,2-*bis*(*p*-chlorophenyl) 1,1,1-trichloroethane (DDT). Br. Med. J. 1945; 1: 865-871

Castello G, Testini G. Gas chromatographic index system for polychlorinated biphenyls: possibilities and limitations. J. Chromatogr. 1997; 787: 215-225

Chapman PM. Current approaches to developing sediment quality criteria. Environ. Toxicol. Chem. 1989; 8: 589-600

Chen ZM, Zabik MJ, Leavitt RA. Comparative study of thin film photodegradative rates for 36 pesticides. Ind. Eng. Chem. Prod. Res. Dev. 1984; 23: 5-11

Chumbley K, Sease J, Strick M, Towell R. Field studies of Steller sea lions 13 (*Eumetopias jubatus*) at Marmot Island, Alaska 1979 through 1994 U.S. Department of 14 Commerce. NOAA Tech. Memo. 1997. NMFS-AFSC-77, 99pp.

Connell DW, Miller G, Anderson S. Chlorohydrocarbon pesticides in the Australian marine environment after banning in the period from the 1970s to 1980s. Mar. Pollut. Bull. 2002; 45: 78-83

Crosby DG, Environmental toxicology and chemistry. Oxford University Press, UK, 1998, pp. 352

De Swart RL, Ross PS, Vedder LJ, Timmermann HH, Heisterkamp S, Loveren HV, Vos

JG, Reijnders PJH, Osterhaus ADME. Impairment of immune function in harbour seals (*Phoca vitulina*) feeding on fish from polluted waters. Ambio 1994; 23: 155-159

De Swart RL, Ross PS, Vos JG, Osterhaus ADME. Impaired Immunity in Harbour Seals (*Phoca vitulina*) Exposed to Bioaccumulated Environmental Contaminants: Review of a Long-Term Feeding Study. Environ. Health Perspect. 1996; 104, Supplement 4: 823-828

DeLong R, Gilmartin WG, Simpson JG. Premature births in California sea lions: association with high organochlorine pollutant residue levels. Science 1973; 21 (181) 1168-1170

Del Toro LD, Heckel G, Camacho-Ibar VF, Schramm Y. California Sea lions (*Zalophus californianus californianus*) have lower chlorinated hydrocarbon contents in northern Baja California, Mexico, than in California USA. Environ. Poll. 2006; 142(1): 83-92

Deutch B, Hansen JC. High human plasma levels of organochlorine compounds in Greenland: Regional differences and lifestyle effects. Dan. Med. Bull. 2000; 47(2): 132-137

Drescher HE, Harms U, Huschenbeth E. Organochlorines and heavy metals in the harbour seal *Phoca vitulina* from the German North Sea Coast. Mar. Biol. 1977; 41(1): 99-106

EPA. 1980a. Ambient water quality criteria for hexachlorocyclohexane. Washington, DC: U.S. Environmental Protection Agency, Criteria and Standards Office. EPA440580054. PB81117657

EPA. 1980c. Manual of analytical methods for the analysis of pesticides in humans and environmental samples. Research Triangle Park, NC: U.S. Environmental Protection Agency, Health Effects Research Laboratory, Environmental Toxicology Division. EPA600880038

EPA. 1981a. Aquatic fate process data for organic priority pollutants. Washington, DC: U.S. Environmental Protection Agency, Office of Water Regulations and Standards. EPA-440/4-81-014

EPA. 1986f. Broad scan analysis of the FY82 national human adipose tissue survey specimens: Volume I -Executive summary. Washington, DC: U.S. Environmental Protection Agency, Office of Toxic Substances. EPA-560/5-86-035

EPA. 1989d. Hydrolysis rate constants for enhancing property-reactivity relationships. Athens, GA: U.S. Environmental Protection Agency. EPA600389063. PB89220479

EPA. 1995c. Toxic chemical release inventory. Reporting form R and instructions. Washington DC: U.S. Environmental Protection Agency, Office of Pollution Prevention and Toxics. EPA745-K-95-051

EPA. 1998k. Method 8270D: Semivolatile organic compounds by gas chromatography/ mass spectrometry (GC/MS). Washington, DC: U.S. Environmental Protection Agency

EPA. 1999k. Universal treatment standards. Washington, DC: U.S. Environmental Protection Agency. 40 CFR 268.48

Ferrero RC, DeMaster DP, Hill PS, Muto MM, Lopez AL. Alaska Marine Mammal Stock assessment. NOAA Technical Memorandum 2000NMFS-AFSC-119, Department of Commerce USA

Fadely B, Fritz L, Ream R, Towell R, Sterling J, Stinchcomb C, Perryman W, Gelatt T. Contrasting Western Steller sea lion and northern fur seal populations trends in Alaska. Alaska Fisheries Science Center (AFSC) Quarterly Report, National Marine Fisheries Service, 2006, pp. 8

Fiscus CH, Baines GA. Food and Feeding Behavior of Steller and California Sea Lions. J. Mammal. 1966; 47(2): 195-200

Fossi MC, Marsili L. Effects of endocrine disruptors in aquatic mammals. Pure Appl. Chem. 2003; 75 (11-12) 2235-2247

Fritz LW, Ferrero RC, Berg RJ. The threatened status of Steller sea lions, *Eumetopias jubatus*, under the Endangered Species Act: effects on Alaska groundfish fisheries management. Mar. Fish. Rev. 1995; 57(2): 14-27

Fritz LW, Ferrero RC. Options in Steller sea lion recovery and groundfish fishery management. Biosphere Conservation 1998; 1(1): 7-19

Gaines T. The acute toxicity of pesticides to rats. Toxicol. Appl. Pharmacol. 1960; 2: 88-99

Gaines T. Acute toxicity of pesticides. Toxicol. Appl. Pharmacol. 1969; 14: 515-534

Garthoff LH, Cerra FE, Marks EM. Blood chemistry alterations in rats after single and multiple gavage administration of polychlorinated biphenyl. Toxicol. Appl. Pharmacol. 1981; 60: 33-44

Geyer H, Sheehan P, Kotzias D. Prediction of ecotoxicological behavior of chemicals: Relationship between physicochemical properties and bioaccumulation of organic chemicals in the mussel *Mytilus edulis*. Chemosphere 1982; 11: 1121-1134

Geyer HJ, Rimkus GG, Scheunert I, Kaune A, Schramm KW, Kettrup A, Zeeman, M, Muir DCG, Hansen LG, Mackay D. Bioaccumulation and Occurrence of Endocrine Disrupting Chemicals (EDCs), Persistent Organic Pollutants (POPs), and Other Organic Compounds in Fish and Other Organisms Including Humans. In: The Handbook of Environmental Chemistry, Vol. 2 Part J Bioaccumulation (ed. B. Beek). Springer-Verlag Berlin Heidelberg 2000

Giesy JP, Kannan KP. Dioxin-like and non-dioxin-like toxic effects of polychlorinated biphenyls (PCBs): Implications for risk assessment. Crit. Rev. Toxicol. 1998; 28(6): 511-569.

Gold B, Brunk G. Metabolism of 1,1,1-trichloro-2,2-bis(*p*-chlorophenyl)ethane and 1,1 dichloro-2,2-bis(*p*-chlorophenyl)ethane in the mouse. Chem. Biol. Interact. 1982; 41: 327-339

Gold B, Brunk G. Metabolism of 1,1,1-trichloro-2,2-bis(*p*-chlorophenyl)ethane (DDT), 1,1-dichloro-2,2-bis(*p*-chlorophenyl)ethane, and 1-chloro-2,2-bis(*p*-chlorophenyl) ethene in the hamster. Cancer. Res. 1983; 43: 2644-2647

Gold B, Brunk G. A mechanistic study of the metabolism of 1,1-dichloro-2,2-bis (*p*-chlorophenyl) ethane (DDD) to 2,2-bis(*p*-chlorophenyl)acetic acid (DDA). Biochem Pharmacol 1984; 33(7): 979-982

Gupta PK. Pesticide exposure-Indian scene. Toxicology 2004; 198: 83–90

Hall AJ., Kalantzi OI. Polybrominated diphenyl ethers (PBDEs) in grey seals during their first year of life - are they thyroid hormone endocrine disrupters? Envrion. Poll. 2003: 126: 29-37

Haller HL, Bartlett PD, Drake NC, Newman MS, Cristol SJ, Eaker CM, Hayes RA, Kilmer GW, Magerlein B, Mueller G, Schneider A, Weatley W. Chemical composition of technical DDT. J. Amer. CHem. Soc. 1945; 67: 1591-1602

Hansen JC. Environmental contaminants and human health in the Arctic. Toxicol. Lett. 2000; 112-113: 119-25. Review

Hansch C, Leo A, Hoekman D. Exploring QSAR: Hydrophobic, electronic, and steric constants. Washington, DC: American Chemical Society, 15; 1995

Harding GC, LeBlanc RJ, Vass WV, Addison RF, Hargrave BT, Pearre S Jr, Dupuis A, Brodie PF. Bioaccumulation of polychlorinated biphenyls (PCBs) in the marine pelagic food web, based on a seasonal study in the southern Gulf of St. Lawrence, 1976-1977. Mar. Chem. 1997; 56(3-4): 145-179

Hardy ML. The toxicity of the commercial polybrominated diphenyl oxide flame retardants: DBDPO, OBDPO, PeBDPO. Organohal. Comp. 2000b; 47: 41-44

Harju M, Bergman A, Roos, A, Haglund P. Determination of atropisomeric and planar polychlorinated biphenyls, their enantiomeric fractions and tissue distribution in grey seals using comprehensive 2D gas chromatography. J. Chromatogr. A 2003; 1019: 127-142

Harwood J, Croxall JP. The assessment of competitions between seals and commercial fisheries in the North Sea and Antarctica. Mar. Mamm. Sci. 1988; 4(1): 13

Hauzenberger I, Perthen-Palmisano B, Hermann M. Lindane. Report presented at the Third Meeting of the POPs Expert Group in Geneva, Switzerland in June 2002

Hawker DW, Connell DW. Bioconcentration of lipophilic compounds by some aquatic organisms. Ecotoxicol Environ. Safety 1986; 11: 184-197

Helle E, Olsson M, Jensen S. PCB Levels Correlated with Pathological Changes in Seal

Uteri. Ambio 1976; 5: 5-6

Helle E. Reproductive trends and occurrence of organochlorines and heavy metals in the Baltic seal population. International Council for the Exploration of the Sea, C.M. 1981E; 37: 13

Helm PA, Bidleman TF, Stern GA, Koczanski, K. Polychlorinated naphthalenes and coplanar polychlorinated biphenyls in beluga whale (*Delphinapterus leucas*) and ringed seal (*Phoca hispida*) from the eastern Canadian Arctic. Environ. Poll. 2002; 119: 69-78

Hobbs KE, Muir DCG. Levels and patterns of persistent organochlorines in minke whale (*Balaenoptera acutorostrata*) stocks from the North Atlantic and European Arctic. Environ. Poll. 2003; 121: 239-252

Holden AV. The accumulation of oceanic contaminants in marine mammals. Symposium on the biology of the seal. vp. 1973. In: Symposium of the biology of seals, 14-17 August 1972, Univ. Guelph, Guelph, Ontario, Canada

Hooper SW, Pettigrew CA, Sayler GS. Ecological fate, effects and prospects for the elimination of environmental polychlorinated biphenyls (PCBs). Environ. Toxicol. Chem 1990; 9(5): 655-667

Hornshaw TC, Aulerich RJ, Johnson HE. Feeding Great Lakes fish to mink: effects on mink and accumulation and elimination of PCBS by mink. J. Toxicol. Environ. Health. 1983; 11(4-6): 933-946

Hutchinson JD, Simmonds MP. Organochlorine contamination in pinnipeds. Rev. Environ. Contam. Toxicol. 1994; 136: 123-168

International Agency for Research on Cancer (IARC). IARC monograph on the evaluation of the carcinogenic risk of chemicals to man: some monomers, plastics and synthetic elastomers, and acrolein. Vol. 19. Lyon: IARC; 1979

Jenssen BM. An Overview of Exposure to, and Effects of, Petroleum Oil and Organochlorine Pollution in Grey Seals (*Halichoerus grypus*). Sci. Tot. Env. 1996; 186(1-2): 109-118

Kajiwara N, Kannan K, Muraoka M, Watanabe M, Takahashi S, Gulland F, Olsen H, Blankenship AL, Jones PD, Tanabe S, Giesy JP. Organochlorine pesticides, polychlorinated biphenyls, and butyltin compounds in blubber and livers of stranded California sea lions, elephant seals, and harbor seals from coastal California, USA. Arch. Environ. Contam. Toxicol. 2001; 41(1): 90-99

Kannan N, Reusch TBH, Schulz-Bull DE, Petrick G, Duinker JC. Chlorobiphenyls: model compounds for metabolism in food chain organisms and their potential use as ecotoxicological stress indicators by the application of the metabolic slope concept. Environ. Sci. Technol. 1995; 29: 1851-1859

Kannan K, Blankenship AL, Jones PD, Giesy JP. Toxicity reference values for the toxic effects of polychlorinated biphenyls to aquatic mammals. Hum. Ecol. Risk. Assess. 2000; 6(1): 181-201

Kannan K, Kajiwara N, Le Boeuf BJ, Tanabe S. Organochlorine pesticides and polychlorinted biphenyls in California sea lions. Envrion. Poll. 2004; 131: 425-434

Kastelein RA, Vaughan N, Wiepkema PR The food consumption of Steller sea lions (*Eumetopias jubatus*). Aquat. Mammals 1990; 15: 137-144

Kawai S, Fukushima M. Relationship between lipid composition and organochlorine levels in the tissues of Striped Dolphin. Mar. Pollut. Bull. 1988; 19(3): 129-133

Keyes MC. The nutrition of pinnipeds. In: RJ Harrison, RC Hubbard, RS Peterson, CE Rice and RJ Shusterman (Eds.). The behavior and physiology of pinnipeds. Appleton-Century-Crofts, New York, NY, 1968, pp. 359-399

Kimbrough RD. Halogentated biphenyls, terphenyls, naphtalenes, dibenzodioxins and related products: Kimbrough RD, Jensen AA, Eds.; Elsevier Science Publishers: Amsterdam. 1989; Chapters 1-2, pp. 3-69

Kleivane L, Skaare JU, Bjorge A, de Ruiter E, Reijnders PJH. Organochlorine pesticide residue and PCBs in Harbor porpoise *(Phocoena phocoena)* incidentally caught in Scandinavian Waters. Environ. Poll. 1995; 89(2): 137-146

Kleivane L, Severinsen T, Skaare JU. Biological transport and mammal to mammal transfer of organochlorines in Arctic fauna. Mar. Pollut. Bull. 2000; 49: 343-357

Krahn MM, Becker PR, Tilbury KL, Stein JE. Organochlorine contaminants in blubber of four seal species. Chemosphere 1997; 34: 2109-2121

Krahn MM. Chlorinated hydrocarbon and DDT analyses of blubber from Steller sea lions from southeast Alaska, Chapter 5.In: Pitcher KW, editor. Steller Sea Lion Recovery Investigations in Alaska 1995–1996 Alaska Department of Fish and Game, 1997

Krahn MM, Beckmen KB, Pitcher PW, Burek KA. Population survey of organochlorine contaminants in Alaskan Steller sea lions. Final Programmatic Report for the NFWF Funded Project October 2. Seattle, WA: National Marine Fisheries Service, 2001

Kucklick JR, Struntza WDJ, Becker PR, York GW, O'Hara TM, Bohonowych JE.

Persistent organochlorine pollutants in ringed seals and polar bears collected from northern Alaska. Sci. Tot. Env. 2002; 287: 45-59

Kutz FW, Wood PH, Bottimore DP. Organochlorine pesticides and polychlorinated biphenyls in human adipose tissue. Rev. Environ. Contam. Toxicol. 1991; 120: 1-82

Lake CA, Lake JL, Haebler R, McKinney R, Boothman WS, Sadove SS. Contaminant levels in harbor seals from the Northeastern United States. Arch. Environ. Contam. Toxicol. 1995; 29: 128-134

Law R, Allchin C, Harwood J. Concentrations of organochlorine compounds in the blubber of seals from eastern and northeastern England, 1988. Mar. Pollut. Bull. 1989; 20: 110-115

Le Boeuf BJ, Giesy JP, Kannan K, Kajiwara N, Tanabe S, Debier C. Organochlorine pesticides in California sea lions revisited. BMC Ecology 2002; 2: 11

Lee JS, Tanabe S, Umino H, Tatsukawa R, Loughlin TR, Calkins DC. Persistent organochlorines in Steller sea lion (*Eumetopias jubatus*) from the bulk of Alaska and the Bering Sea, 1976-1981. Mar. Pollut. Bull. 1996; 32(7): 535-544

Li YF, Cai DJ, Singh A. Technical hexachlorocyclohexane use trends in China and their impact on the environment. Arch. Environ. Contam. Toxicol. 1998; 35: 688-690

Lieberg CP, Bacon CE, Burns SA, Jarman WM, Le Boeuf BJ. DDT in Claifornia sea lions: A follow-up study after twenty years. Mar. Pollut. Bull. 1995; 30: 744-745

Linder RE, Gaines TB, Kimbrough RD. The effect of polychlorinated biphenyls on rat reproduction. Food. Cosmet. Toxicol. 1974; 12: 63-77

Loganathan BG, Tanabe S, Tanaka H, Watanabe S, Miyazaki N, Amano M, Tatsukawa

R. Comparison of organochlorine residue levels in the striped dolphin from the western North Pacific 1978-79 and 1986. Mar. Pollut. Bull. 1990; 21: 435-439

Loughlin TR, Rugh DJ, Fiscus CH. Northern sea lion distribution and abundance: 1956-1980. J. Wildl. Manage. 1984; 48: 729-740

Loughlin TR. Using the phylogeographic method to identify Steller sea lion stocks. Pp. 329-341, *In* A. Dizon, S. J. Chivers, and W. Perrin (eds.), Molecular genetics of marine mammals, incorporating the proceedings of a workshop on the analysis of genetic data to address problems of stock identity as related to management of marine mammals. Soc. Mar. Mammal., Spec. Rep. No. 3 1997

Loughlin TR, York AE. An accounting of the sources of Steller sea lion, *Eumetopia jubatus*, mortality. Mar. Fish. Rev. 2000; 64: 40-45

Loughlin TR, Castellini MA, Ylitalo G. Spatial aspects of organochlorine contamination in northern fur seal tissues. Mar. Pollut. Bull 2002; 44: 1024-1034

Lowry LF, Frost KJ. Biological interactions between marine mammals and commercial fisheries in the Bering Sea, p. 277-283. *In* J. R. Beddington, RJH. Beverton, and DM Lavigne (eds.), Marine mammals and fisheries. George Allen and Unwin (Publishers), London, UK 1985

Manz M, Wenzel KD, Dietze U, Schüürmann G. Persistent organic pollutants in agricultural soils of central Germany. Sci. Total Environ. 2001; 277: 187-198

Marine Mammal Commission. Annual report of the Marine Mammal Commission, calendar year 2002. Report to Congress. 2003. 253 pp. + iv. Available from the Marine Mammal Commission, Bethesda, Maryland

Martineau D, Béland P, Desjardins C, Legacé A. Levels of organochlorine chemicals I tissues of beluga whales (*Delphinapterus leucas*) from the St. Lawrence estuary, Quebec, Canada. Arch. Environ. Contam. Toxicol. 1987; 16: 137-147

Matheson LJ, Tratnyek PG. Reductive dehalogenation of chlorinated methanes by iron metal. Environ. Sci. Technol. 1994; 28: 2045-2053

Mathisen OA, Baade RT, Lopp RJ. Breeding Habits, Growth and Stomach Contents of the Steller Sea Lion in Alaska. J. Mammal. 1962; 43(4): 469-477

McCarty LS, Borgert CJ. Review of the toxicity of chemical mixtures containing at least one organochlorine. Regul. Toxicol. Pharmacol. 2006; 45: 104-118

Mees J, Reijnders PJH. The harbour seal, *Phoca vitulina*, in the Oosterschelde: decline and possibilities for recovery. Hydrobiologia 1994; 282-283: 547-555

Merrick RL, Loughlin TR, Calkins DG. Decline in abundance of the northern sea lion, *Eumetopias jubatus*, in Alaska, 1956-86. Fishery Bulletin. 1987; 85(2): 351-365

Merrick RL, Chumbley MK, Byrd GV. Diet diversity of Steller sea lions (*Eumetopias jubatus*) and their population decline in Alaska: a potential relationship. Can. J. Fish. Aquat. Sci./J. can. sci. halieut. Aquat. 1997; 54(6): 1342-1348

Metcalfe PJ. Organolead compounds in the atmosphere. Analyt. Proc. 1989; 26: 134-136

Metcalf RL. Insect Control Technology. In: Kroschwitz J, Howe-Grant M, eds. Kirk-Othmer encyclopedia of chemical technology. Volume 14. New York, NY: John Wiley and Sons, Inc., 1995. pp 524-602

Miao XS, Swenson C, Woodward LA, Li QX. Distribution of polychlorinated biphenyls in marine species from French Shoals, North Pacific Ocean. Sci. Total. Environ. 2000; 257:17-28

Miles JRW, Tu CM, Harris CR. Degradation of heptachlor epoxide and heptachlor by a mixed culture of soil microorganisms. J. Econ. Entomol. 1971; 64: 839-841

Miller KA, Assunacao MGL, Dangerfield NJ, Bandiera SM, Ross PR. Assessmnet of cytochrome P450 1A in harbor seals (*Phoca vitulina*) using a minimally-invasive biopsy approach. Mar. Environ. Res. 2005; 60: 153-169

Miller KP, Borgeest C, Greenfeld C, Tomic D, Flaws JA. In utero effects of chemicals on reproductive tissues in females. Toxicol. Appl. Pharmacol. 2004; 198: 111 –131

Minh NH, Minh TB, Kajiwara N, Kunisue T, Subramanian A, Iwata H, Tana TS, Baburajendran R, Karuppiah S, Viet PH, Tuyen BC, Tanabe S. Contamination by Persistent Organic Pollutants in Dumping Sites of Asian Developing Countries: Implication of Emerging Pollution Sources. Arch. Environ. Contam. Toxicol. 2006; 50: 474–481

Mortensen P, Bergman A, Bignert A, Hansen H-J, Härkönen T, and Olsson M. Prevalence of skull lesions in harbor seals (*Phoca vitulina*) in Swedish and Danish museum collections: 1835-1988. Ambio 1992; 21: 520-524

Mössner S, Ballschmitter K. Marine mammals as global pollution indicators for organochlorines. Chemosphere 1997; 34(5-7): 1285-1296

Muir DCG, Nostrom RJ, Simon M. Organochlorine contaminants in the Arctic marine food chains: accumulation of specific polychlorinated biphenyls and chlordane-related compounds. Environ. Sci. Technol. 1988; 22: 1071-1079

Muir DC, Wagemann R, Hargrave BT, Thomas DJ, Peakall DB, Norstrom RJ. Arctic marine ecosystem contamination. Sci. Total Environ. 1992; 122: 75-134

Muir DCG, Koczanaski K, Rosenberg B, Beland P. Persistent organochlorines in Beluga whales (Delphinapterus leucas) from the St. Lawrence River estuary - II.

Temporal trends, 1982-1994. Environ. Poll. 1996; 93: 235-245

Muir D, Braune B, DeMarch B, Norstrom R, Wagemann R, Lockhart L, Hargrave B, Bright D, Addison R, Payne P, Reimer K. Spatial and temporal trends and effects of contaminants in the Canadian Arctic marine ecosystem: a review. Sci. Tot. Env. 1999; 230: 83-144

Muir D, Savinova T. Bioaccumulation of PCBs and chlorinated pesticides in seals, fishes and invertebrates form the White Sea, Russia. Sci. Tot. Env. 2003; 306: 111-131

Muir DCG, Shearer RG, Oostdam JV, Donaldson SG, Furgal C. Contaminants in Canadian arctic biota and implications for human health: Conclusions and knowledge gaps. Sci. Total Environ. 2005; 351-352: 539-546

Nakata H, Tanabe S, Tatsukawa R, Amano M, Miyazaki N, Petrov EA. Persistent organochlorine residues and their accumulation kinetics in Baikal seal (*Phoca sibirica*) from Lake Baikal, Russia. Environ. Sci. Technol. 1995; 29: 2877-2885

National Marine Fisheries Service. Recovery plan for the Steller sea lion *(Eumetopias jubatus)*. National Marine Fisheries Service. NMFS 1992; 120 pp.

Norstrom RJ, Muir DCG. Organochlorine Contamination in the Arctic Marine Ecosystem: Implications for Marine Mammals. Proc. Atlantic Coast Contaminants Workshop 2000, Endocrine Disruptors in the Marine Environment: Impacts on Marine Wildlife and Human Health, Bar Harbor, Maine, June 22-25, 2000, pp. 38-43

O'Hara TM, Krahn MM, Boyd D, Becker PR, Philo LM. Organochlorine contaminant levels in Eskimo harvested bowhead whales of arctic Alaska. J. Wildl. Dis. 1999; 35(4): 741-52

Okeke BC, Siddique T, Arbestain MC, Frankenberger WT. Biodegradation of gamma-hexachlorocyclohexane (lindane) and alpha-hexachlorocyclohexane in water and a soil slurry by a *Pandoraea species*. J. Agric. Food Chem. 2002; 50(9): 2548-2555

Olsson M, Karlsson B, Ahnland E. Diseases and environmental contaminants in seals from the Baltic and the Swedish west coast. Sci. Tot. Env. 1994; 154: 217-227

Overton E, Byrne C, McFall J, Antoine S. Tissue levels of trace organic and heavy metal pollutants in subsistence harvested bowhead whales, *Balena mysticetus*. In T.F. Albert (Ed.), Fifth Conference on the Biology of the Bowhead Whale (*Balaena mysticetus*), Anchorage, 1990

Pasanen M. The expression and regulation of drug metabolism in human placenta. Adv. Drug Deliv. Rev. 1999; 38(1): 81-97

Patterson DG Jr., Todd GD, Turner WE. Levels of non-ortho-substituted (coplanar), mono and di-ortho-substituted polychlorinated biphenyls, dibenzo-p-dioxins, and dibenzofurans in human serum and adipose tissue. Environ. Health Perspect. Suppl. 1994; 102(1): 195-204

Perez MA, Loughlin TR. Incidental catch of marine mammals by foreign and JV trawl vessels in the U.S. EEZ of the North Pacific. U.S. Dep. Commer., NOAA Tech. Rep. NMFS 104, 1991; 57 pp.

Piatt JF, Anderson P. Responses of common murres to the Exxon Valdez oil spill and long-term changes in the Gulf of Alaska marine ecosystem. Am. Fish. Soc. Symp. 1996; 18: 720-737

Pitcher KW, Calkins DG. Reproductive Biology of Steller Sea Lions in the Gulf of

Alaska. J. Mammal. 1981; 62(3): 599-605

Pitcher KW. Prey of the Steller sea lion, *Eumetopias jubatus*, in the Gulf of Alaska. Fish. Bull. 1981; 79: 467-472

Quinn M, Allen E. Changes in incidence of and mortality from breast cancer in England and Wales since introduction of screening. United Kingdom Association of Cancer Registries. Br. Med. J. 1995; 311: 1391-1395

Rasmussen JB, Rowan DJ, Lean DRS, Carey JH. Food chain structure in Ontario Lake determines PCB levels in lake trout (*Salvelinus namaycush*) and other pelagic fish. Can. J. Fish. Aquat. Sci. 1990; 47(10): 2030-2038

Reijnders PJH. Organochlorine and Heavy Metal Residues in Harbour Seals from the Wadden Sea and Their Possible Effects on Reproduction Netherlands J. Sea Res. 1980; 14(1): 30-65

Reijnders PJH. Reproductive failure in common seals feeding on fish from polluted coastal waters. Nature 1986; 324: 456-457

Rosen DAS, Trites AW. Pollock and the decline of Steller sea lions: testing the junk-food hypothesis Can. J. Zool. 2000; 78(7): 1243-1250

Ross P, De Swart R, Addison R, Van Loveren H, Vos J, Osterhaus A. Contaminant-induced immunotoxicity in harbour seals: Wildlife at risk? Toxicology 1996; 112(2): 157-169

Ross P. The role of immunotoxic environmental contaminants in facilitating the emergence of infectious diseases in marine mammals. HERA 2002; 8(2): 277-292

Ross G. The public health implications of PCBs in the environment. Ecotoxicol. Environ.

Saf. 2004; 59(3): 275-291

Rubinstein NI, Gilliam WT, Gregory NR. Dietary accumulation of PCBs from a contaminated sediment source by demersal fish (*Leiostomus xanthurus*). Aquat. Toxicol. 1984; 5: 331-342

Safe S, Polychlorinated biphenyls (PCBs) and polybrominated biphenyls (PBBS): biochemistry, toxicology and mechanism of action. CRC Crit. Rev. Toxicol. 1984; 13: 319-395

Safe S. Polychlorinated biphenyls (PCBs), Dibenzo-*p*-dioxins (PCDDs), dibenzofurans (PCDFs), and related compounds: Environmental and mechanistic considerations which support the developement of toxic equivalency factors (TEFs). CRC Crit. Rev. Toxicol. 1990; 21(1): 51-88

Savitskii IV. The basis for determining safe permissible concentrations of hexachlorobenzene and pentachloronitrobenzene in the air. Chemical Abstracts 1965; 63: 8952

Sease JL, Strick JM, Merrick RL, Lewis J.P. Aerial and land-based surveys of Steller sea lions (*Eumetopias jubatus*) in Alaska, June and July 1996. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-99, 1999, pp. 43

Severinsen T, Skaare JU. Spartial distribution of persistent organochlorines in ringed seal (*Phoca hispida*) blubber. Mar. Environ. Res. 2000; 49: 291-302

Shaw SD, Brenner D, Bourakovsky A, Mahaffey CA, Perkins CR. Polychlorinated biphenyls and chlorinated pesticides in harbor seals (*Phoca vitulina concolor*) from the northwestern Atlantic coast. Mar. Pollut. Bull 2005; 1069-1084

Sinclair EH, Zeppelin TZ. Seasonal and spatial differences in diet in the western stock of

Steller sea lions (*Eumetopias jubatus*). J. Mammal. 2002; 83: 973-990

Storelli MM, Marcotrigiano GO. Environmental contamination in bottlenose dolphin (*Tursiops truncatus*): Relationship between levels of metals, methylmercury, and organochlorine compounds in an adult female, her neonate, and a calf. Bull. Environ. Contam. Toxicol. 2000B; 64: 333-340

Storelli MM, Marcotrigiano GO. Levels and congener pattern of polychlorinated biphenyls in the blubber of the Mediterranean bottlenose dolphins (*Tursiops truncates*). Environ. Int. 2003; 28(7): 559-565

Storr-Hansen E, Spliid H. Coplanar polychlorinated biphenyl congener levels and patterns and the identification of separate populations of harbor seals (*Phoca vitulina*) in Denmark. Arch. Environ. Contam. Toxicol. 1993; 24(1): 44-58

Strachan WMJ, Burniston DA, Williamson M, Bohdanowicz. Spatial differences in persistent organochlorine pollutants concentrations between the Bering and Chukchi Seas (1993). Mar. Pollut. Bull. 2001; 43(1-6); 132-142

Swain WR. Human health consequences of consumption of fish contaminated with organochlorine compounds. Aquat. Toxicol. 1988; 11(3-4): 357-377

Tanabe SS, Tatsucawa R, Miura S. Variations of organochlorine residues with age and sex in Antarctic minke whale. Mem. Natle. Inst. Polar. Res. 1985; 44: 174-181

Tanabe S, Sung JK, Choi DY, Baba N, Kiyota M, Yoshida K, Tatsukawa R. Persistent organochlorine residues in northern fur seal from the Pacific coast of Japan since 1971. Environ. Poll. 1994; 85: 305-314

Teramitsu I, Yamamoto Y, Chiba I, Iwata H, Tanabe S, Fujise Y, Kazusaka A, Akahori F, Fujiay S. Identification of novel cytochrom P450 1a genes from five marine mammal species. Aquat. Toxicol. 2000; 51: 145-153

Trites AW, Christensen V, Pauly D. Competition between fisheries and marine mammals for prey and primary production in the Pacific Ocean. J. Northw. Atl. Fish. Sci. 1997; 22: 173-187

Troisi GM, Mason CF. Cytochromes P450, P420 & mixed function oxidases as biomarkers of polychlorinated biphenyl (PCB) exposure in harbor seals (*Phoca vitulina*). Chemosphere 1997; 35(9): 1933-1946

Tashiro S, Matsumura F. Metabolism of trans-nonachlor and related chlordane components in rat and man. Arch. Environ. Contam. Toxicol. 1978; 7: 113-127

UNEP (United Nations Environment Programme): UNEP 1999 report

United States Department of Interior (USDOI), FWS. 2006. Fish and Wildlife Service ESA (endagered Species Act) Section 7, 2005

Van den Berg M, Birnbaum L, Bosveld ATC, Brunström B, Cook P, Feeley M, Giesy P, Hanberg A, Hasegawa R, Kennedy SW, Kubiak T, Larsen JC, van Leeuwen FXR, Liem AKD, Nolt C, Peterson RE, Poellinger L, Safe S, Schrenk D, Tillitt D, Tysklinf M, Younes M, Wærn F, Zacharewski T. Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. Environ. Health. Perspect. 1998; 106(12): 775-792

Varanasi U, Stein JE, Reichert WL, Tilbury KL, Krahn MM, Chan SL. Chlorinated and aromatic hydrocarbons in bottom sediments, fish and marine mammals in US coastal waters: laboratory and field studies of metabolism and accumulation. In: CH Walker and DR Livingstone (eds). Persistent Pollutants in Marine Ecosystems. Oxford: Pergamon Press, 1992; pp. 83-115

Veith GD, Macek KJ, Petrocelli SR, Carrol J. 1980. An evaluation of using partition coefficients and water solubility to estimate bioconcentration factors for organic chemicals in fish, in Aquatic Toxicology (J.G. Eaton, P.R.Parrish, and A.C. Hendricks, eds.), ASTM, Philadelphia, PA. STP 707, pp.116-129

Vos JG, Luster MI. Immune alterations. In halogenated biphenyls, terphenyls, naphthalenes, dibenzodioxins, and related products (R. D. Kimbrough and A. A. Jensen, Eds.), pp. 295-322. Elsevier Science Pub. 1989

Vos JG, Dybing E, Greim HA, Ladefoged O, Lambre C, Tarazona J V, Brandt I, Vethaak AD. Health effects of endocrine-disrupting chemicals on wildlife, with special reference to the European situation. Crit. Rev. Toxicol. 2000; 30: 71-133

Walter H, Consolaro F, Gramatica P, Scholze M, Altenburger R. Mixture toxicity of priority pollutants at no observed effect concentrations (NOECs) Ecotoxicol. 2002; 11: 299-310

Wania F, Mackay D. Global fractionation and cold condensation of low volatile organochlorine compounds in polar regions. Ambio 1993; 22: 10-18

Wania F, Mackay D. Tracking the distribution of persistent organic pollutants. Environ. Sci. Technol. 1996; 30: 390A-396A

Wania F. On the origin of elevated levels of persistent chemicals in the environment. Environ. Sci. Pollut. Res. 1999; 6: 11-19

Wang D, Atkinson S, Hoover-Miller A, Lee SE, Li, QX. Organochlorines in harbor seal (*Phoca vitulina*) tissues from the northern Gulf of Alaska. Environ. Poll. 2007; 146(1): 268-80

Watanabe M, Kannan K, Takahashi A, Loganathan BG, Odell DK, Tanabe S, Giesy JP.

Polychlorinated biphenyls, organochlorine pesticides tris (4-chlorophenyl) methane, and tris(4-chlororphenyl)methanol in livers of small cetaceans stranded along Florida coastal waters USA. Environ. Toxicol. Chem. 2000; 19: 1566-1574

White RD, Shea D, Schlezinger JJ, Hahn ME, Stegeman JJ. In vitro metabolism of polychlorinated biphenyl congeners by beluga whale (*Delphinapterus leucas*) and pilot whale (*Globicephala melas*) and relationship to cytochrome P450 expression. Comp. Biochem. Physiol. C. Toxicol. Pharmacol. 2000; 126(3): 267-284

Willcox MK, Woodward L, Ylitalo G, Buzitis J, Atkinson S, Li QX. Survey for organochlorines in the free-ranging Hawaiian monk seal (*Monachus schauinslandi*) at French Frigate Shoals, North Pacific Ocean. Sci. Total Environ. 2004; 322(1-3): 81-93

Williams S. Official methods of analysis. Association of Official Analytical Chemists, Inc., Arlington, VA. 1984

Williams RJ. Practical applications of marine bioremediation. Curr. Opin. Biotechnol. 1998; 9: 300-304

Winship AJ, Trites AW, Rosen DAS. A bioenergetic model for estimating the food requirements of Steller sea lions Eumetopias jubatus in Alaska, USA. Mar. Ecol. Prog. Ser. 2002; 229: 291-312

Wolkers J, Burkow IC, Monshouwer M, Lydersen C, Dahle S, Witkamp RF. Cytochrom P450-mediated enzyme activity and polychlorinated biphenyl accumulation in Harp seal (*Phoca groenlandica*). Mar. Environ. Res. 1999; 48: 59-71

World Health Organization (WHO). 1975 evaluations of some pesticide residues in food.

World Health Organization. Pest. Res. Ser. 1976; (5): 267-271, pp. 396

World Health Organization. DDT and its derivatives. Environmental health criteria 9. Geneva: World Health Organization, United Nations Environment Programme, 1979.

World Health Organization (WHO). Vector Control for Malaria and Other Mosquitoborne Diseases. Report of a WHO Study Group. Technical Report Series, no. 857. 1995; Geneva

York AE. The population dynamics of northern sea lions, 1975-1985. Mar. Mamm. Sci. 1994; 10: 38-51

Zhang G, Parker A, House A, Mai B, Li X, Kang Y. Sedimentary records of DDT and HCH in the Pearl River Delta, South China. Environ. Sci. Technol. 2002; 36: 3671-3677

Zhang Q, Liang X, Chen J, Lu P, Yediler A, Kettrup A. Correct identification of polychlorinated biphenyls in temperature-programmed GC with ECD detection. Anal. Bioanal. Chem. 2002; 374: 93-102

APPENDIX A: GLOSSARY

(Modified after ATSDR reports 1998, 2000, 20002 and 2005)

Absorption - The taking up of liquids by solids, or of gases by solids or liquids.

Adsorption - The adhesion in an extremely thin layer of molecules (as of gases, solutes, or liquids) to the surfaces of solid bodies or liquids with which they are in contact.

Adverse effect - any change in body function or the structures of cells that can lead to disease or health problems.

Bioconcentration Factor (BCF) - The quotient of the concentration of a chemical in aquatic organisms at a specific time or during a discrete time period of exposure divided by the concentration in the surrounding water at the same time or during the same period.

BZ number - A system of sequential numbers for the 209 PCB congeners introduced in 1980 by Ballschmiter and Zell that identifies a given congener simply and precisely. Also referred to as congener, IUPAC, or PCB number.

Carcinogen - A chemical capable of inducing cancer.

Congener - A single, unique, well-defined chemical compound in the PCB category. The name of the congener specifies the total number of chlorine substituents and the position of each chlorine.

Congener number - A system of sequential numbers for the 209 PCB congeners introduced in 1980 by Ballschmiter and Zell that identifies a given congener simply and precisely. Also referred to as BZ, PCB, or IUPAC number.

Developmental Toxicity - The occurrence of adverse effects on the developing organism that may result from exposure to a chemical prior to conception (either parent), during prenatal development, or postnatally to the time of sexual maturation. Adverse developmental effects may be detected at any point in the life span of the organism.

Embryotoxicity and Fetotoxicity - Any toxic effect on the conceptus as a result of prenatal exposure to a chemical; the distinguishing feature between the two terms is the stage of development during which the insult occurs. The terms, as used here, include malformations and variations, altered growth, and *in utero* death.

Genotoxicity - A specific adverse effect on the genome of living cells that, upon the duplication of affected cells, can be expressed as a mutagenic, clastogenic or carcinogenic event because of specific alteration of the molecular structure of the genome.

Half-life - A measure of rate for the time required to eliminate one half of a quantity of a chemical from the body or environmental media.

Immunologic Toxicity - The occurrence of adverse effects on the immune system that may result from exposure to environmental agents such as chemicals.

Immunological Effects - Functional changes in the immune response.

IUPAC number - A system of sequential numbers for the 209 PCB congeners introduced in 1980 by Ballschmiter and Zell that identifies a given congener simply and precisely. Also referred to as BZ, congener, or PCB number.

Lethal Dose(50) (LD50) - The dose of a chemical which has been calculated to cause death in 50% of a defined experimental animal population.

Mortality - Death; mortality rate is a measure of the number of deaths in a population during a specified interval of time.

Neurotoxicity - The occurrence of adverse effects on the nervous system following exposure to a chemical.

Octanol-Water Partition Coefficient (Kow) -The equilibrium ratio of the concentrations of a chemical between *n*-octanol and water, in dilute solution.

Organic Carbon Distribution Coefficient (Koc) - An estimate of a persistent organic pollutant adsorbing to soil or sediment particles.

Pesticide - General classification of chemicals specifically developed and produced for use in the control of agricultural and public health pests.

Risk - The possibility or chance that some adverse effect will result from a given exposure to a chemical.

Risk Factor - An aspect of personal behavior or lifestyle, an environmental exposure, or an inborn or inherited characteristic, that is associated with an increased occurrence of disease or other health-related event or condition.

Teratogen - A chemical that causes structural defects that affects the development of an organism.

Toxicokinetic - The study of the absorption, distribution and elimination of toxic compounds in the living organism.

Xenobiotic - Any chemical that is foreign to the biological system.

Appendix B: PCB original data for Steller sea lion blubber, liver, kidney and placenta tissues from chapter 2

1. PCB concentrations (dry weights) of Steller sea lion blubber tissues

Location Sample ID Cl-No.	ng/g dry weight IUPAC No.	Tatitlek SSL-2 (1) Blubber	Tatitlek SSL-3 Blubber	Tatitlek SSL-4 Blubber	Tatitlek SSL-2 (5) Blubber	Tatitlek SSL-1 Blubber	Median	Interquartile Range	Range Min	Range Max	Average	StDev
2	7+9	0.0	0.0	0.0	2.2	0.0	0.0	0.0	0.0	2.2	0.4	1.0
2	13	0.0	0.0	12.5	23.4	10.7	10.7	12.5	0.0	23.4	9.3	9.8
3	16+32	0.0	0.0	0.0	2.2	0.0	0.0	0.0	0.0	2.2	0.4	1.0
3	22	0.0	0.0	0.0	3.0	0.0	0.0	0.0	0.0	3.0	0.6	1.3
3	34	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	26	0.0	0.0	0.0	2.2	0.0	0.0	0.0	0.0	2.2	0.4	1.0
3	28	0.0	6.7	10.8	19.7	0.0	6.7	10.8	0.0	19.7	7.4	8.3
3	31	0.0	0.0	0.0	0.0	10.1	0.0	0.0	0.0	10.1	2.0	4.5
4	41+71	0.0	0.0	0.0	2.5	2.9	0.0	2.5	0.0	2.9	1.1	1.5
4	44	0.0	6.7	7.6	8.8	5.1	6.7	2.5	0.0	8.8	5.7	3.4
4	47+48+75	0.0	0.0	0.0	4.1	0.0	0.0	0.0	0.0	4.1	0.8	1.8
4	52+73	8.2	7.7	15.0	15.0	0.0	8.2	7.3	0.0	15.0	9.2	6.2
4	53	0.0	0.0	0.0	2.1	0.0	0.0	0.0	0.0	2.1	0.4	1.0
4	62	0.0	3.4	5.0	3.7	5.8	3.7	1.6	0.0	5.8	3.6	2.2
4	66+80+93+95	7.6	10.0	16.6	13.2	20.7	13.2	6.6	7.6	20.7	13.6	5.2
4	67+100	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	70	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	74	8.1	16.0	25.2	14.5	25.8	16.0	10.7	8.1	25.8	17.9	7.5
4	81+87+111	0.0	0.0	2.9	4.5	0.0	0.0	2.9	0.0	4.5	1.5	2.1
5	82	13.2	11.7	24.1	9.1	13.4	13.2	1.8	9.1	24.1	14.3	5.8
5	83	0.0	0.0	0.0	4.8	0.0	0.0	0.0	0.0	4.8	1.0	2.1
5	84	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	90+101	62.4	79.0	89.1	40.7	100.9	79.0	26.7	40.7	100.9	74.4	23.5
5	99	0.0	2.6	11.5	4.6	12.1	4.6	8.9	0.0	12.1	6.1	5.4
5	105	7.3	24.0	7.3	5.6	0.0	7.3	1.7	0.0	24.0	8.8	9.0
5	109	0.0	0.0	7.1	2.5	18.0	2.5	7.1	0.0	18.0	5.5	7.6
5	118	76.9	115.1	120.0	54.1	0.0	76.9	61.0	0.0	120.0	73.2	49.2
5	119	0.0	97.1	147.5	102.9	185.2	102.9	50.4	0.0	185.2	106.5	69.5
5	122+131+142	0.0	28.3	33.5	18.3	18.6	18.6	10.0	0.0	33.5	19.7	12.8
5	124+147	0.0	41.6	96.3	53.9	0.0	41.6	53.9	0.0	96.3	38.4	40.5
6	128	35.8	11.2	24.2	10.0	32.4	24.2	21.2	10.0	35.8	22.7	11.9
6	130	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	132	0.0	0.0	0.0	4.4	0.0	0.0	0.0	0.0	4.4	0.9	2.0
6	134	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	135+144	0.0	0.0	0.0	0.0	75.4	0.0	0.0	0.0	75.4	15.1	33.7
6	136	0.0	0.0	9.2	22.7	3.9	3.9	9.2	0.0	22.7	7.2	9.5
6	138+163+164	137.0	130.6	74.3	31.9	73.8	74.3	56.8	31.9	137.0	89.5	44.0
6	141	0.0	9.4	22.4	7.7	17.4	9.4	9.7	0.0	22.4	11.4	8.7
6	146	0.0	0.0	25.6	10.7	0.0	0.0	10.7	0.0	25.6	7.3	11.3
6	149	11.8	8.5	25.1	20.0	111.4	20.0	13.3	8.5	111.4	35.4	43.0
6	151	3.5	0.0	0.0	0.0	7.6	0.0	3.5	0.0	7.6	2.2	3.4
6	153	177.9	175.5	175.8	66.9	183.6	175.8	2.4	66.9	183.6	156.0	49.9
6	154	0.0	164.1	282.3	158.5	164.0	164.0	5.5	0.0	282.3	153.8	100.5
6	157+201	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.4	0.5	1.1
6	158	0.0	0.0	0.0	0.0	11.9	0.0	0.0	0.0	11.9	2.4	5.3
6	183	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	165	0.0	0.0	0.0	0.0	8.2	0.0	0.0	0.0	8.2	1.6	3.7
7	170+190	14.2	19.3	33.0	9.0	28.4	19.3	14.2	9.0	33.0	20.8	9.9
7	171+202	6.1	0.0	7.0	0.0	5.4	5.4	6.1	0.0	7.0	3.7	3.4
7	172	6.2	0.0	0.0	0.0	3.2	0.0	3.2	0.0	6.2	1.9	2.8
7	174	9.0	3.6	10.0	4.9	8.3	8.3	4.1	3.6	10.0	7.2	2.7
7	175	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	177	3.6	0.0	6.1	3.3	4.5	3.6	1.2	0.0	6.1	3.5	2.2
7	178	4.0	4.9	17.8	4.4	0.0	4.4	0.9	0.0	17.8	6.2	6.7
7	180	61.7	37.3	100.9	37.6	96.9	61.7	59.3	37.3	100.9	66.9	30.9
7	187	0.0	0.0	53.3	26.8	0.0	0.0	26.8	0.0	53.3	16.0	23.8
7	193	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8	194	5.3	0.0	7.5	0.0	4.9	4.9	5.3	0.0	7.5	3.5	3.4
8	195+208	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.7	0.3	0.7
8	196+203	3.1	0.0	6.2	0.0	3.9	3.1	3.9	0.0	6.2	2.6	2.7
8	199	4.5	0.0	9.6	0.0	7.4	7.4	7.4	0.0	9.6	4.3	4.3
8	200	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sum	ng/g dry wt	671.3	1014.3	1522.1	836.3	1281.6	1014.3	342.9	671.3	1522.1	1065.1	341.2

Location Sample ID Cl-No.	ng/g dry weight IUPAC No.	St. Paul SNPSLS9802 Blubber	St. Paul SNPSLS9901 Blubber	St. Paul SNPSLS2000-04 Blubber	St. Paul SP-01-00-EJ Blubber	St. Paul SNPSLS9808 Blubber	St. Paul Karin (SP-01-01-EJ) Blubber	Median	Interquartile Range	Range Min	Range Max	Average	StDev
2	7+9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2	13	12.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	12.2	2.0	5.0
3	16+32	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	22	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	34	0.0	0.0	39.7	0.0	0.0	0.0	0.0	0.0	0.0	39.7	6.6	16.2
3	26	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	28	4.6	7.1	0.0	0.0	0.0	0.0	0.0	3.4	0.0	7.1	2.0	3.1
3	31	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	41+71	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	44	0.0	9.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	9.9	1.6	4.0
4	47+48+75	0.0	6.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.6	1.1	2.7
4	52+73	6.9	28.3	1.6	13.8	0.0	38.4	10.4	21.7	0.0	38.4	14.8	15.5
4	53	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	62	2.6	8.2	0.0	0.0	0.0	0.0	0.0	2.0	0.0	8.2	1.8	3.3
4	66+80+93+95	8.9	21.6	4.7	11.2	9.7	41.5	10.4	10.0	4.7	41.5	16.3	13.6
4	67+100	0.0	0.0	3.3	0.0	0.0	0.0	0.0	0.0	0.0	3.3	0.5	1.3
4	70	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	74	18.6	31.1	4.7	12.7	4.8	49.0	15.7	21.2	4.7	49.0	20.2	17.2
4	81+87+111	1.5	5.0	0.0	0.0	0.0	0.0	0.0	1.1	0.0	5.0	1.1	2.0
5	82	10.1	24.8	5.3	7.7	0.0	51.7	8.9	15.2	0.0	51.7	16.6	19.1
5	83	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	84	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	90+101	264.5	106.9	36.0	132.8	128.1	0.0	117.5	77.9	0.0	264.5	111.4	92.0
5	99	2.2	4.3	0.0	0.0	0.0	0.0	0.0	1.6	0.0	4.3	1.1	1.8
5	105	84.2	7.8	2.3	0.0	23.8	17.7	12.7	18.6	0.0	84.2	22.6	31.5
5	109	0.0	4.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.2	0.7	1.7
5	118	150.3	114.9	33.1	62.3	78.5	0.0	70.4	65.5	0.0	150.3	73.2	54.4
5	119	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	122+131+142	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	124+147	20.3	0.0	0.0	31.9	0.0	0.0	0.0	15.3	0.0	31.9	8.7	14.0
6	128	11.0	23.3	4.8	5.4	10.9	0.0	8.1	6.0	0.0	23.3	9.2	8.1
6	130	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	132	3.5	8.4	0.0	0.0	0.0	0.0	0.0	2.6	0.0	8.4	2.0	3.4
6	134	4.0	33.7	0.0	0.0	0.0	0.0	0.0	3.0	0.0	33.7	6.3	13.5
6	135+144	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	136	0.0	11.0	0.0	10.5	0.0	0.0	0.0	7.9	0.0	11.0	3.6	5.6
6	138+163+164	242.7	92.4	18.3	44.0	127.1	138.5	109.8	79.5	18.3	242.7	110.5	79.7
6	141	13.3	19.9	0.0	0.0	0.0	0.0	0.0	9.9	0.0	19.9	5.5	8.8
6	146	7.4	35.9	0.0	9.0	0.0	31.9	8.2	24.3	0.0	35.9	14.0	15.9
6	149	8.0	28.8	4.0	11.4	12.9	45.6	12.1	16.0	4.0	45.6	18.4	15.8
6	151	0.0	13.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.8	2.3	5.7
6	153	300.5	195.2	230.8	83.5	220.1	207.1	213.6	30.0	83.5	300.5	206.2	70.5
6	154	137.1	233.5	0.0	93.7	0.0	0.0	46.9	126.3	0.0	233.5	77.4	96.1
6	157+201	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	158	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	183	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	165	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	170+190	14.7	37.9	3.5	4.4	3.9	0.0	4.2	8.6	0.0	37.9	10.7	14.2
7	171+202	4.1	9.0	3.4	0.0	0.0	0.0	1.7	3.9	0.0	9.0	2.7	3.6
7	172	0.0	6.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.6	1.1	2.7
7	174	4.6	17.3	4.7	0.0	0.0	29.6	4.6	13.0	0.0	29.6	9.3	11.7
7	175	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	177	1.9	11.3	0.0	0.0	0.0	11.7	1.0	9.0	0.0	11.7	4.2	5.7
7	178	7.4	18.1	3.2	0.0	0.0	8.1	5.3	7.1	0.0	18.1	6.1	6.8
7	180	41.7	124.6	16.0	28.8	34.4	103.6	38.1	57.9	16.0	124.6	58.2	44.6
7	187	28.4	78.2	0.0	17.6	0.0	0.0	8.8	25.7	0.0	78.2	20.7	30.5
7	193	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8	194	0.0	0.0	3.5	0.0	0.0	0.0	0.0	0.0	0.0	3.5	0.6	1.4
8	195+208	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8	196+203	0.0	0.0	0.0	0.0	0.0	7.6	0.0	0.0	0.0	7.6	1.3	3.1
8	199	0.0	0.0	6.4	0.0	0.0	0.0	0.0	0.0	0.0	6.4	1.1	2.6
8	200	8.7	0.0	0.0	0.0	6.9	0.0	0.0	5.2	0.0	8.7	2.6	4.1
Sum	ng/g dry wt	1425.6	1379.7	429.0	580.9	660.9	781.9	721.4	629.3	429.0	1425.6	876.3	423.7

2. PCB concentrations (lipid weights) of Steller sea lion blubber tissuesl

Location Sample ID Cl-No.	ng/g lipid weight IUPAC No.	Tatitlek SSL-3 Blubber	Tatitlek SSL-4 Blubber	Tatitlek SSL-2 (5) Blubber	Median	Interquartile Range	Range Min	 Max	Average	StDev
2	7+9	0.0	0.0	5.9	0.0	2.9	0.0	5.9	2.0	3.4
2	13	0.0	24.0	63.3	24.0	31.6	0.0	63.3	29.1	31.9
3	16+32	0.0	0.0	6.1	0.0	3.0	0.0	6.1	2.0	3.5
3	22	0.0	0.0	8.1	0.0	4.1	0.0	8.1	2.7	4.7
3	34	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	26	0.0	0.0	6.0	0.0	3.0	0.0	6.0	2.0	3.5
3	28	15.9	20.8	53.2	20.8	18.7	15.9	53.2	30.0	20.3
3	31	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	41+71	0.0	0.0	6.7	0.0	3.3	0.0	6.7	2.2	3.9
4	44	16.0	14.7	23.8	16.0	4.6	14.7	23.8	18.2	4.9
4	47+48+75	0.0	0.0	11.1	0.0	5.6	0.0	11.1	3.7	6.4
4	52+73	18.4	28.9	40.5	28.9	11.1	18.4	40.5	29.2	11.1
4	53	0.0	0.0	5.8	0.0	2.9	0.0	5.8	1.9	3.3
4	62	8.1	9.6	10.0	9.6	0.9	8.1	10.0	9.2	1.0
4	66+80+93+95	23.9	32.0	35.6	32.0	5.9	23.9	35.6	30.5	6.0
4	67+100	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	70	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	74	38.0	48.5	39.2	39.2	5.2	38.0	48.5	41.9	5.7
4	81+87+111	0.0	5.5	12.1	5.5	6.1	0.0	12.1	5.9	6.1
5	82	27.8	46.4	24.5	27.8	11.0	24.5	46.4	32.9	11.8
5	83	0.0	0.0	12.9	0.0	6.5	0.0	12.9	4.3	7.5
5	84	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	90+101	188.1	171.3	110.0	171.3	39.0	110.0	188.1	156.5	41.1
5	99	6.1	22.1	12.4	12.4	8.0	6.1	22.1	13.5	8.0
5	105	57.1	13.9	15.0	15.0	21.6	13.9	57.1	28.7	24.6
5	109	0.0	13.6	6.6	6.6	6.8	0.0	13.6	6.7	6.8
5	118	274.1	230.8	146.2	230.8	64.0	146.2	274.1	217.0	65.1
5	119	231.2	283.6	278.1	278.1	26.2	231.2	283.6	264.3	28.8
5	122+131+142	67.4	64.4	49.4	64.4	9.0	49.4	67.4	60.4	9.6
5	124+147	99.1	185.1	145.6	145.6	43.0	99.1	185.1	143.3	43.0
6	128	26.5	46.6	26.9	26.9	10.0	26.5	46.6	33.4	11.5
6	130	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	132	0.0	0.0	11.9	0.0	5.9	0.0	11.9	4.0	6.9
6	134	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	135+144	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	136	0.0	17.7	61.4	17.7	30.7	0.0	61.4	26.4	31.6
6	138+163+164	311.0	142.9	86.1	142.9	112.4	86.1	311.0	180.0	116.9
6	141	22.5	43.0	20.9	22.5	11.1	20.9	43.0	28.8	12.3
6	146	0.0	49.3	29.0	29.0	24.6	0.0	49.3	26.1	24.8
6	149	20.2	48.3	54.0	48.3	16.9	20.2	54.0	40.9	18.1
6	151	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	153	417.9	338.0	180.9	338.0	118.5	180.9	417.9	312.3	120.6
6	154	390.6	542.8	428.5	428.5	76.1	390.6	542.8	454.0	79.2
6	157+201	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	158	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	183	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	165	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	170+190	45.9	63.5	24.3	45.9	19.6	24.3	63.5	44.6	19.6
7	171+202	0.0	13.5	0.0	0.0	6.7	0.0	13.5	4.5	7.8
7	172	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	174	8.7	19.2	13.4	13.4	5.3	8.7	19.2	13.7	5.3
7	175	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	177	0.0	11.7	8.9	8.9	5.8	0.0	11.7	6.9	6.1
7	178	11.6	34.1	11.8	11.8	11.3	11.6	34.1	19.2	13.0
7	180	88.9	194.1	101.7	101.7	52.6	88.9	194.1	128.2	57.4
7	187	0.0	102.5	72.5	72.5	51.2	0.0	102.5	58.3	52.7
7	193	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8	194	0.0	14.4	0.0	0.0	7.2	0.0	14.4	4.8	8.3
8	195+208	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8	196+203	0.0	11.8	0.0	0.0	5.9	0.0	11.8	3.9	6.8
8	199	0.0	18.5	0.0	0.0	9.3	0.0	18.5	6.2	10.7
8	200	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sum	ng/g lipid weight	2415.0	2927.2	2260.2	2415.0	333.5	2260.2	2927.2	2534.1	971.7

Location Sample ID Cl-No.	ng/g lipid weight IUPAC No.	St. Paul SNPSLS9802 Blubber	St. Paul SNPSLS9901 Blubber	St. Paul SNPSLS2000-04 Blubber	St. Paul SP-01-00-EJ Blubber	St. Paul SNPSLS9808 Blubber	St. Paul Karin (SP-01-01-EJ) Blubber	Median	Interquartile Range	Range Min	 Max	Average	StDev
2	7+9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2	13	23.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	23.4	3.9	9.6
3	16+32	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	22	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	34	0.0	0.0	127.9	0.0	0.0	0.0	0.0	32.0	0.0	127.9	21.3	52.2
3	26	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	28	8.8	12.9	0.0	0.0	0.0	0.0	0.0	3.2	0.0	12.9	3.6	5.8
3	31	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	41+71	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	44	0.0	18.0	0.0	0.0	0.0	0.0	0.0	4.5	0.0	18.0	3.0	7.3
4	47+48+75	0.0	12.0	0.0	0.0	0.0	0.0	0.0	3.0	0.0	12.0	2.0	4.9
4	52+73	13.3	51.4	5.1	26.6	0.0	89.3	15.8	29.0	0.0	89.3	31.0	34.0
4	53	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	62	5.0	14.9	0.0	0.0	0.0	0.0	0.0	3.7	0.0	14.9	3.3	6.0
4	66+80+93+95	17.0	39.3	15.0	21.5	21.0	96.6	21.3	6.5	15.0	96.6	35.1	31.4
4	67+100	0.0	0.0	10.5	0.0	0.0	0.0	0.0	2.6	0.0	10.5	1.8	4.3
4	70	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	74	35.8	56.5	15.3	24.5	10.4	114.0	19.9	18.4	10.4	114.0	42.7	38.6
4	81+87+111	2.9	9.1	0.0	0.0	0.0	0.0	0.0	2.3	0.0	9.1	2.0	3.7
5	82	19.5	45.1	17.2	14.9	0.0	120.3	16.0	13.0	0.0	120.3	36.2	43.7
5	83	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	84	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	90+101	508.6	194.4	116.0	255.4	278.5	0.0	224.9	86.3	0.0	508.6	225.5	171.8
5	99	4.1	7.7	0.0	0.0	0.0	0.0	0.0	1.9	0.0	7.7	2.0	3.3
5	105	161.9	14.1	7.3	0.0	51.7	41.1	10.7	18.1	0.0	161.9	46.0	60.2
5	109	0.0	7.7	0.0	0.0	0.0	0.0	0.0	1.9	0.0	7.7	1.3	3.1
5	118	289.0	209.0	106.7	119.8	170.7	0.0	145.3	63.8	0.0	289.0	149.2	98.5
5	119	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	122+131+142	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	124+147	39.1	0.0	0.0	61.3	0.0	0.0	0.0	15.3	0.0	61.3	16.7	26.9
6	128	21.1	42.4	15.5	10.3	23.7	0.0	19.6	14.1	0.0	42.4	18.8	14.3
6	130	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	132	6.7	15.3	0.0	0.0	0.0	0.0	0.0	3.8	0.0	15.3	3.7	6.3
6	134	7.8	61.2	0.0	0.0	0.0	0.0	0.0	15.3	0.0	61.2	11.5	24.6
6	135+144	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	136	0.0	20.0	0.0	20.2	0.0	0.0	10.0	20.1	0.0	20.2	6.7	10.4
6	138+163+164	466.7	168.1	58.9	84.7	276.2	322.2	126.4	116.9	58.9	466.7	229.5	153.5
6	141	25.5	36.3	0.0	0.0	0.0	0.0	0.0	9.1	0.0	36.3	10.3	16.3
6	146	14.3	68.3	0.0	17.3	0.0	74.1	8.7	29.3	0.0	74.1	28.5	32.8
6	149	15.3	52.4	13.0	21.8	28.0	105.9	24.9	14.5	13.0	105.9	39.4	35.5
6	151	0.0	25.2	0.0	0.0	0.0	0.0	0.0	6.3	0.0	25.2	4.2	10.3
6	153	577.9	354.9	744.5	160.6	478.5	481.6	436.7	258.7	160.6	744.5	466.3	198.0
6	154	263.6	424.5	0.0	180.3	0.0	0.0	90.1	241.3	0.0	424.5	144.7	176.9
6	157+201	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	158	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	183	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	165	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	170+190	28.3	69.0	11.1	8.5	8.4	0.0	9.8	17.1	0.0	69.0	20.9	25.3
7	171+202	7.9	16.3	11.0	0.0	0.0	0.0	5.5	12.3	0.0	16.3	5.9	7.0
7	172	0.0	11.9	0.0	0.0	0.0	0.0	0.0	3.0	0.0	11.9	2.0	4.9
7	174	8.8	31.4	15.3	0.0	0.0	68.7	7.6	19.3	0.0	68.7	20.7	26.3
7	175	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	177	3.7	20.6	0.0	0.0	0.0	27.2	0.0	5.2	0.0	27.2	8.6	12.1
7	178	14.2	32.9	10.2	0.0	0.0	18.6	5.1	15.9	0.0	32.9	12.7	12.5
7	180	80.2	226.6	51.6	53.4	74.9	240.9	65.1	58.3	51.6	240.9	121.6	87.7
7	187	54.5	142.1	0.0	33.9	0.0	0.0	16.9	60.9	0.0	142.1	38.4	55.6
7	193	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8	194	0.0	0.0	11.2	0.0	0.0	0.0	0.0	2.8	0.0	11.2	1.9	4.6
8	195+208	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8	196+203	0.0	0.0	0.0	0.0	0.0	17.6	0.0	0.0	0.0	17.6	2.9	7.2
8	199	0.0	0.0	20.5	0.0	0.0	0.0	0.0	5.1	0.0	20.5	3.4	8.4
8	200	16.7	0.0	0.0	0.0	15.0	0.0	0.0	3.7	0.0	16.7	5.3	8.2
Sum	ng/g lipid weight	2741.5	2508.5	1384.0	1117.1	1436.8	1818.4	1627.6	938.8	1117.1	2741.5	1834.4	1545.8

3. PCB concentrations (dry weights) of Steller sea lion liver tissues

Location Sample ID Cl-No.	ng/g dry weight IUPAC No.	Tatitlek SSL-2 (1) Liver	Tatitlek SSL-3 Liver	Tatitlek SSL-4 Liver	Tatitlek SSL-1 Liver	Tatitlek SSL-2 (5) Liver	Median	Interquartile Range	Range Min	Range Max	Average	StDev
2	4+10	0.0	0.0	0.0	0.0	6.2	0.0	0.0	0.0	6.2	1.2	2.8
2	7+9	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.2	0.3
2	13	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2	14	0.0	0.0	0.0	0.0	8.1	0.0	0.0	0.0	8.1	1.6	3.6
3	16+32	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	18	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	19	10.0	6.3	5.4	0.0	21.1	6.3	4.6	0.0	21.1	8.6	7.9
3	20	0.0	0.0	0.0	0.0	15.6	0.0	0.0	0.0	15.6	3.1	7.0
3	22	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	34	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	24+27	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	26	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	28	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	29+54	0.0	0.0	0.0	0.0	4.0	0.0	0.0	0.0	4.0	0.8	1.8
3	31	4.1	3.8	3.6	3.7	4.2	3.8	0.4	3.6	4.2	3.9	0.3
3	35	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	40	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	41+71	48.3	7.4	44.7	14.9	26.1	26.1	29.8	7.4	48.3	28.3	18.0
4	44	3.8	0.0	0.0	0.0	3.4	0.0	3.4	0.0	3.8	1.4	2.0
4	46	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	47+48+75	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	52+73	3.4	0.0	2.8	0.0	3.1	2.8	3.1	0.0	3.4	1.9	1.7
4	53	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	66+80+93+95	4.9	5.9	3.7	5.3	5.6	5.3	0.6	3.7	5.9	5.1	0.9
4	67+100	4.2	0.0	0.0	3.6	0.0	0.0	3.6	0.0	4.2	1.6	2.2
4	69	3.9	2.9	3.4	2.7	5.4	3.4	1.0	2.7	5.4	3.7	1.1
4	70	4.4	3.7	4.0	3.6	5.0	4.0	0.7	3.6	5.0	4.1	0.6
4	74	0.0	4.8	0.0	5.5	0.0	0.0	4.8	0.0	5.5	2.1	2.8
5	82	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	83	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	84	8.7	7.5	7.4	7.8	19.1	7.8	1.2	7.4	19.1	10.1	5.0
5	90+101	2.3	0.0	1.2	2.3	2.3	2.3	1.2	0.0	2.3	1.6	1.0
5	92	31.4	6.3	25.9	10.5	12.7	12.7	15.4	6.3	31.4	17.4	10.8
5	99	0.0	0.0	0.0	0.0	40.6	0.0	0.0	0.0	40.6	8.1	18.2
5	105	2.9	3.1	2.8	3.0	3.3	3.0	0.1	2.8	3.3	3.0	0.2
5	109	6.5	3.9	4.3	3.8	3.3	3.9	0.5	3.3	6.5	4.4	1.2
5	110	7.7	4.8	3.8	4.3	2.9	4.3	1.0	2.9	7.7	4.7	1.8
5	115+116+117	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	118	2.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.6	0.5	1.2
5	119	5.2	3.0	3.8	2.8	0.0	3.0	0.9	0.0	5.2	2.9	1.9
5	122+131+142	0.0	0.0	0.0	0.0	2.8	0.0	0.0	0.0	2.8	0.6	1.3
5	124+147	2.8	0.0	0.0	0.0	2.5	0.0	2.5	0.0	2.8	1.1	1.4
5	125	4.1	3.0	3.4	2.8	2.3	3.0	0.6	2.3	4.1	3.1	0.7
6	128	0.0	3.0	3.5	2.7	3.4	3.0	0.7	0.0	3.5	2.5	1.5
6	130	7.5	5.6	5.5	6.0	5.7	5.7	0.4	5.5	7.5	6.1	0.8
6	132	3.2	2.5	2.7	2.5	2.6	2.6	0.2	2.5	3.2	2.7	0.3
6	136	0.0	0.0	0.0	0.0	3.1	0.0	0.0	0.0	3.1	0.6	1.4
6	138+163+164	3.2	3.1	3.0	2.9	2.8	3.0	0.1	2.8	3.2	3.0	0.1
6	146	10.2	10.2	9.5	10.9	10.2	10.2	0.1	9.5	10.9	10.2	0.5
6	149	14.2	10.0	10.8	10.1	6.4	10.1	0.8	6.4	14.2	10.3	2.8
6	151	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	153	5.9	3.9	5.6	4.0	2.6	4.0	1.8	2.6	5.9	4.4	1.4
6	154	4.6	3.8	3.3	3.3	3.5	3.5	0.5	3.3	4.6	3.7	0.6
6	157+201	12.5	3.1	4.5	0.0	2.5	3.1	2.0	0.0	12.5	4.5	4.7
6	158	0.0	0.0	0.0	2.5	2.8	0.0	2.5	0.0	2.8	1.1	1.4
6	183	3.0	4.5	4.2	5.2	3.8	4.2	0.7	3.0	5.2	4.1	0.8
6	165	4.2	2.9	3.1	2.6	2.7	2.9	0.4	2.6	4.2	3.1	0.6
7	170+190	3.7	4.0	3.8	2.9	4.1	3.8	0.4	2.9	4.1	3.7	0.5
7	172	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	173	0.0	0.0	0.0	0.0	2.6	0.0	0.0	0.0	2.6	0.5	1.2
7	174	4.5	3.6	3.6	3.1	4.2	3.6	0.6	3.1	4.5	3.8	0.6
7	175	3.1	3.8	5.7	3.2	2.8	3.2	0.7	2.8	5.7	3.7	1.1
7	177	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	178	2.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.8	0.6	1.3
7	179	3.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.1	0.6	1.4
7	180	3.3	2.7	3.1	0.0	0.0	2.7	3.1	0.0	3.3	1.8	1.7
7	187	14.9	3.8	3.4	3.0	4.1	3.8	0.7	3.0	14.9	5.8	5.1
7	191	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	193	3.3	0.0	4.2	0.0	0.0	0.0	3.3	0.0	4.2	1.5	2.1
8	194	3.2	2.9	3.1	2.9	3.7	3.1	0.3	2.9	3.7	3.1	0.3
8	195+208	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8	196+203	3.8	0.0	3.8	0.0	6.0	3.8	3.8	0.0	6.0	2.7	2.6
8	197	12.5	7.1	9.3	6.8	8.8	8.8	2.3	6.8	12.5	8.9	2.3
8	199	4.3	3.1	4.4	0.0	5.1	4.3	1.3	0.0	5.1	3.4	2.0
8	200	0.0	15.0	0.0	0.0	5.1	0.0	5.1	0.0	15.0	4.0	6.5
8	205	0.0	3.2	2.9	0.0	4.4	2.9	3.2	0.0	4.4	2.1	2.0
9	207	0.0	0.0	3.0	0.0	0.0	0.0	0.0	0.0	3.0	0.6	1.3
Sum	ng/g dry weight	292.4	168.9	226.3	151.4	302.6	226.3	123.5	151.4	302.6	228.3	69.1

Location Sample ID Cl-No.	ng/g dry weight IUPAC No.	St. Paul SNPSLS9901 Liver	St. Paul SNPSLS9802 Liver	St. Paul SNPSLS2000-04 Liver	St. Paul SNPSLS9808 Liver	St. Paul Karin (SP-01-01-EJ) Liver	Median	Interquartile Range	Range Min	Range Max	Average	StDev
2	4+10	0.0	0.0	0.0	2.5	0.0	0.0	0.0	0.0	2.5	0.5	1.1
2	7+9	6.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.6	1.3	3.0
2	13	0.0	0.0	3.0	0.0	0.0	0.0	0.0	0.0	3.0	0.6	1.3
2	14	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	16+32	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	18	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	19	0.0	7.7	28.4	23.5	12.0	12.0	15.8	0.0	28.4	14.3	11.6
3	20	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	22	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	34	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	24+27	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	26	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	28	0.0	0.0	0.0	0.0	2.7	0.0	0.0	0.0	2.7	0.5	1.2
3	29+54	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	31	0.0	3.5	0.0	0.0	4.7	0.0	3.5	0.0	4.7	1.6	2.3
3	35	4.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.0	0.8	1.8
4	40	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	41+71	6.0	0.0	0.0	7.6	0.0	0.0	6.0	0.0	7.6	2.7	3.8
4	44	2.1	0.0	3.9	5.7	0.0	2.1	3.9	0.0	5.7	2.4	2.5
4	46	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	47+48+75	2.3	0.0	0.0	6.8	0.0	0.0	2.3	0.0	6.8	1.8	3.0
4	52+73	2.5	0.0	3.0	0.0	3.1	2.5	3.0	0.0	3.1	1.7	1.6
4	53	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	66+80+93+95	3.1	4.7	4.5	4.3	4.0	4.3	0.5	3.1	4.7	4.1	0.6
4	67+100	2.8	5.2	4.0	3.6	3.4	3.6	0.6	2.8	5.2	3.8	0.9
4	69	2.4	3.3	0.0	2.6	3.2	2.6	0.8	0.0	3.3	2.3	1.3
4	70	3.1	4.4	4.1	3.8	4.5	4.1	0.6	3.1	4.5	4.0	0.6
4	74	4.5	3.1	6.2	4.3	6.2	4.5	1.9	3.1	6.2	4.8	1.3
5	82	0.0	0.0	0.0	2.3	0.0	0.0	0.0	0.0	2.3	0.5	1.0
5	83	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	84	4.5	10.3	8.0	9.6	8.1	8.1	1.7	4.5	10.3	8.1	2.2
5	90+101	0.0	2.4	2.5	3.0	2.2	2.4	0.3	0.0	3.0	2.0	1.2
5	92	5.2	4.4	7.0	5.3	6.2	5.3	1.0	4.4	7.0	5.6	1.0
5	99	0.0	0.0	0.0	0.0	3.0	0.0	0.0	0.0	3.0	0.6	1.4
5	105	2.3	3.1	2.8	3.0	3.0	3.0	0.2	2.3	3.1	2.8	0.3
5	109	3.7	4.6	4.3	3.6	4.7	4.3	0.9	3.6	4.7	4.2	0.5
5	110	3.3	4.1	4.5	4.4	3.4	4.1	1.0	3.3	4.5	3.9	0.6
5	115+116+117	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	118	0.0	0.0	2.5	2.4	2.5	2.4	2.5	0.0	2.5	1.5	1.3
5	119	3.3	2.7	3.5	3.3	4.3	3.3	0.3	2.7	4.3	3.4	0.6
5	122+131+142	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	124+147	0.0	0.0	0.0	0.0	0.9	0.0	0.0	0.0	0.9	0.2	0.4
5	125	3.1	3.1	3.5	3.0	2.6	3.1	0.1	2.6	3.5	3.1	0.3
6	128	2.8	5.3	4.5	3.9	12.1	4.5	1.4	2.8	12.1	5.7	3.7
6	130	4.4	9.2	5.7	6.9	5.9	5.9	1.2	4.4	9.2	6.4	1.8
6	132	2.4	3.3	2.7	2.7	2.7	2.7	0.1	2.4	3.3	2.7	0.3
6	136	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	138+163+164	2.7	3.0	2.9	3.0	2.9	2.9	0.1	2.7	3.0	2.9	0.1
6	146	6.8	15.9	10.0	13.3	10.8	10.8	3.3	6.8	15.9	11.4	3.4
6	149	7.5	10.5	11.6	10.5	10.8	10.5	0.3	7.5	11.6	10.2	1.6
6	151	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	153	4.4	3.9	4.6	3.9	4.7	4.4	0.7	3.9	4.7	4.3	0.4
6	154	3.1	6.5	3.7	4.5	4.0	4.0	0.9	3.1	6.5	4.4	1.3
6	157+201	3.9	6.7	6.5	7.1	0.0	6.5	2.8	0.0	7.1	4.9	3.0
6	158	0.0	3.2	0.0	3.7	0.0	0.0	3.2	0.0	3.7	1.4	1.9
6	183	3.8	7.5	9.2	9.1	19.5	9.1	1.7	3.8	19.5	9.8	5.8
6	165	2.7	3.0	2.9	2.7	2.8	2.8	0.2	2.7	3.0	2.8	0.1
7	170+190	3.7	5.9	2.7	3.4	2.8	3.4	0.9	2.7	5.9	3.7	1.3
7	172	0.0	2.4	4.3	0.0	7.2	2.4	4.3	0.0	7.2	2.8	3.1
7	173	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	174	3.5	8.2	4.7	2.8	7.1	4.7	3.6	2.8	8.2	5.2	2.3
7	175	3.4	6.0	12.6	8.7	20.2	8.7	6.5	3.4	20.2	10.2	6.5
7	177	0.0	3.0	0.0	0.0	0.0	0.0	0.0	0.0	3.0	0.6	1.3
7	178	0.0	0.0	2.4	0.0	2.6	0.0	2.4	0.0	2.6	1.0	1.4
7	179	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	180	0.0	3.7	3.6	2.8	0.0	2.8	3.6	0.0	3.7	2.0	1.9
7	187	2.6	4.8	6.5	6.7	10.0	6.5	1.8	2.6	10.0	6.1	2.7
7	191	3.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.2	0.6	1.4
7	193	0.0	5.1	4.1	2.9	5.3	4.1	2.2	0.0	5.3	3.5	2.2
8	194	2.7	3.1	9.0	0.0	3.2	3.1	0.4	0.0	9.0	3.6	3.3
8	195+208	0.0	4.0	0.0	0.0	0.0	0.0	0.0	0.0	4.0	0.8	1.8
8	196+203	0.0	2.7	3.7	3.8	3.7	3.7	1.0	0.0	3.8	2.8	1.6
8	197	5.9	10.4	7.0	9.6	7.8	7.8	2.6	5.9	10.4	8.1	1.8
8	199	2.8	3.9	5.5	3.2	7.0	3.9	2.3	2.8	7.0	4.5	1.7
8	200	2.4	3.4	2.4	2.6	0.0	2.4	0.2	0.0	3.4	2.2	1.3
8	205	2.6	3.0	0.0	0.0	0.0	0.0	2.6	0.0	3.0	1.1	1.6
9	207	2.7	5.2	3.3	3.1	0.0	3.1	0.6	0.0	5.2	2.9	1.9
Sum	ng/g dry weight	145.0	219.1	231.7	225.6	237.6	225.6	12.6	145.0	237.6	211.8	38.0

4. PCB concentrations (lipid weights) of Steller sea lion liver tissues

Location Sample ID Cl-No.	ng/g lipid weight IUPAC No.	Tatitlek SSL-2 (1) Liver	Tatitlek SSL-3 Liver	Tatitlek SSL-4 Liver	Tatitlek SSL-1 Liver	Tatitlek SSL-2 (5) Liver	Median	Interquartile Range	Range Min	 Max	Average	StDev
2	4+10	0.0	0.0	0.0	0.0	162.6	0.0	0.0	0.0	162.6	32.5	72.7
2	7+9	0.0	12.6	0.0	0.0	0.0	0.0	0.0	0.0	12.6	2.5	5.6
2	13	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2	14	0.0	0.0	0.0	0.0	212.9	0.0	0.0	0.0	212.9	42.6	95.2
3	16+32	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	18	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	19	109.5	104.2	86.3	0.0	550.9	104.2	23.2	0.0	550.9	170.2	217.4
3	20	0.0	0.0	0.0	0.0	407.8	0.0	0.0	0.0	407.8	81.6	182.4
3	22	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	34	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	24+27	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	26	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	28	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	29+54	0.0	0.0	0.0	0.0	103.4	0.0	0.0	0.0	103.4	20.7	46.3
3	31	45.2	63.4	57.3	64.8	111.0	63.4	7.5	45.2	111.0	68.3	25.0
3	35	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	40	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	41+71	528.4	121.7	712.3	259.2	683.4	528.4	424.3	121.7	712.3	461.0	261.3
4	44	41.2	0.0	0.0	0.0	90.1	0.0	41.2	0.0	90.1	26.3	39.9
4	46	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	47+48+75	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	52+73	37.2	0.0	45.3	0.0	80.8	37.2	45.3	0.0	80.8	32.7	34.0
4	53	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	66+80+93+95	53.7	97.6	58.3	93.0	145.4	93.0	39.4	53.7	145.4	89.6	37.0
4	67+100	46.3	0.0	0.0	63.3	0.0	0.0	46.3	0.0	63.3	21.9	30.6
4	69	43.0	47.9	55.0	47.4	140.0	47.9	7.6	43.0	140.0	66.7	41.3
4	70	48.1	61.2	63.9	62.6	130.6	62.6	2.6	48.1	130.6	73.3	32.6
4	74	0.0	79.8	0.0	95.9	0.0	0.0	79.8	0.0	95.9	35.1	48.5
5	82	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	83	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	84	95.3	123.6	118.8	136.0	498.6	123.6	17.2	95.3	498.6	194.5	170.7
5	90+101	25.5	0.0	18.4	40.4	61.2	25.5	22.0	0.0	61.2	29.1	23.1
5	92	343.8	103.8	413.3	182.9	331.1	331.1	161.0	103.8	413.3	275.0	127.3
5	99	0.0	0.0	0.0	0.0	1062.7	0.0	0.0	0.0	1062.7	212.5	475.2
5	105	32.2	50.4	44.7	52.1	87.5	50.4	7.4	32.2	87.5	53.4	20.6
5	109	70.9	63.9	69.1	65.8	85.7	69.1	5.1	63.9	85.7	71.1	8.6
5	110	83.9	79.0	60.3	75.1	74.5	75.1	4.5	60.3	83.9	74.6	8.8
5	115+116+117	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	118	28.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	28.8	5.8	12.9
5	119	56.7	48.7	59.8	49.2	0.0	49.2	8.0	0.0	59.8	42.9	24.4
5	122+131+142	0.0	0.0	0.0	0.0	74.3	0.0	0.0	0.0	74.3	14.9	33.2
5	124+147	30.3	0.0	0.0	0.0	64.9	0.0	30.3	0.0	64.9	19.0	28.8
5	125	44.7	49.8	54.4	49.2	60.7	49.8	5.2	44.7	60.7	51.7	6.1
6	128	0.0	49.7	56.6	47.4	89.2	49.7	9.2	0.0	89.2	48.6	31.9
6	130	82.3	93.1	88.2	104.6	148.6	93.1	16.4	82.3	148.6	103.4	26.6
6	132	35.3	40.4	42.8	43.0	66.9	42.8	2.6	35.3	66.9	45.7	12.3
6	136	0.0	0.0	0.0	0.0	81.6	0.0	0.0	0.0	81.6	16.3	36.5
6	138+163+164	34.8	50.4	47.3	50.9	73.4	50.4	3.5	34.8	73.4	51.4	14.0
6	146	111.7	168.4	152.0	189.7	265.6	168.4	37.7	111.7	265.6	177.5	56.9
6	149	155.1	164.9	172.1	176.1	167.8	167.8	7.2	155.1	176.1	167.2	8.0
6	151	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	153	65.0	63.7	89.8	69.7	66.8	66.8	4.7	63.7	89.8	71.0	10.7
6	154	50.3	63.1	52.6	56.6	92.4	56.6	10.5	50.3	92.4	63.0	17.1
6	157+201	136.2	51.1	71.1	0.0	64.9	64.9	20.0	0.0	136.2	64.7	48.8
6	158	0.0	0.0	0.0	42.8	73.3	0.0	42.8	0.0	73.3	23.2	33.6
6	183	32.6	74.3	66.9	89.9	99.8	74.3	23.0	32.6	99.8	72.7	25.9
6	165	46.0	47.9	50.0	45.5	70.7	47.9	4.0	45.5	70.7	52.0	10.6
7	170+190	40.0	66.5	60.7	50.8	107.7	60.7	15.8	40.0	107.7	65.1	25.8
7	172	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	173	0.0	0.0	0.0	0.0	69.0	0.0	0.0	0.0	69.0	13.8	30.9
7	174	49.4	58.9	57.0	53.6	109.8	57.0	5.2	49.4	109.8	65.7	24.9
7	175	33.6	61.9	90.4	56.1	73.6	61.9	17.5	33.6	90.4	63.1	21.1
7	177	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	178	31.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	31.0	6.2	13.9
7	179	34.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	34.1	6.8	15.3
7	180	36.2	44.1	50.0	0.0	0.0	36.2	44.1	0.0	50.0	26.1	24.3
7	187	163.1	62.5	54.6	51.6	107.3	62.5	52.8	51.6	163.1	87.8	47.7
7	191	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	193	36.1	0.0	67.4	0.0	0.0	0.0	36.1	0.0	67.4	20.7	30.4
8	194	35.0	47.0	48.9	80.7	97.1	48.9	3.6	35.0	97.1	55.7	23.9
8	195+208	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8	196+203	41.8	0.0	60.0	0.0	157.6	41.8	60.0	0.0	157.6	51.9	64.7
8	197	137.2	116.7	148.9	117.6	229.3	137.2	31.3	116.7	229.3	149.9	46.4
8	199	46.7	51.0	70.2	0.0	133.9	51.0	23.5	0.0	133.9	60.4	48.5
8	200	0.0	248.0	0.0	0.0	133.4	0.0	133.4	0.0	248.0	76.3	112.0
8	205	0.0	52.4	45.9	0.0	114.5	45.9	52.4	0.0	114.5	42.6	47.2
9	207	0.0	0.0	47.8	0.0	0.0	0.0	0.0	0.0	47.8	9.6	21.4
Sum	ng/g lipid weight	3198.5	2784.1	3608.5	2633.4	7914.2	3198.5	824.5	2633.4	7914.2	4027.7	2205.6

Location Sample ID Cl-No.	ng/g lipid weight IUPAC No.	St. Paul SLS9901 Liver	St. Paul SSLS9802SNP Liver	St. Paul SNPSLS2000-04 Liver	St. Paul SLS9808 Liver	St. Paul Karin (SP-01-01-ZJ) Liver	Median	Interquartile Range	Range Min	 Max	Average	StDev
2	4+10	0.0	0.0	0.0	48.6	0.0	0.0	0.0	0.0	48.6	9.7	21.7
2	7+9	117.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	117.9	23.6	52.7
2	13	0.0	0.0	49.2	0.0	0.0	0.0	0.0	0.0	49.2	9.8	22.0
2	14	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	16+32	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	18	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	19	0.0	232.3	469.7	458.7	243.0	243.0	226.4	0.0	469.7	280.8	193.6
3	20	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	22	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	34	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	24+27	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	26	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	28	0.0	0.0	0.0	0.0	54.7	0.0	0.0	0.0	54.7	10.9	24.5
3	29+54	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	31	0.0	105.4	0.0	0.0	95.1	0.0	95.1	0.0	105.4	40.1	55.0
3	35	71.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	71.3	14.3	31.9
4	40	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	41+71	107.1	0.0	0.0	147.5	0.0	0.0	107.1	0.0	147.5	50.9	71.2
4	44	37.7	0.0	64.3	112.1	0.0	37.7	64.3	0.0	112.1	42.8	47.3
4	46	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	47+48+75	40.5	0.0	0.0	132.5	0.0	0.0	40.5	0.0	132.5	34.6	57.5
4	52+73	43.8	0.0	50.4	0.0	62.3	43.8	50.4	0.0	62.3	31.3	29.3
4	53	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	66+80+93+95	55.9	142.5	73.7	83.7	80.4	80.4	10.0	55.9	142.5	87.2	32.7
4	67+100	50.0	157.5	66.8	69.8	69.9	69.8	3.1	50.0	157.5	82.8	42.6
4	69	42.2	97.8	0.0	51.0	64.4	51.0	22.2	0.0	97.8	51.1	35.5
4	70	55.2	131.7	67.4	74.2	92.2	74.2	24.7	55.2	131.7	84.2	29.8
4	74	79.9	92.3	102.8	82.9	125.1	92.3	19.8	79.9	125.1	96.6	18.3
5	82	0.0	0.0	0.0	45.1	0.0	0.0	0.0	0.0	45.1	9.0	20.2
5	83	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	84	80.7	309.3	131.9	187.8	164.9	164.9	55.8	80.7	309.3	174.9	85.2
5	90+101	0.0	71.9	42.0	59.3	45.4	45.4	17.4	0.0	71.9	43.7	27.2
5	92	93.0	131.3	116.3	104.0	125.8	116.3	21.8	93.0	131.3	114.1	15.7
5	99	0.0	0.0	0.0	0.0	61.7	0.0	0.0	0.0	61.7	12.3	27.6
5	105	41.4	93.0	46.6	59.2	59.9	59.2	13.4	41.4	93.0	60.0	20.1
5	109	66.1	137.6	71.0	70.9	94.8	71.0	23.9	66.1	137.6	88.1	29.9
5	110	58.9	122.7	74.2	85.3	68.1	74.2	17.2	58.9	122.7	81.8	24.7
5	115+116+117	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	118	0.0	0.0	40.9	46.4	50.1	40.9	46.4	0.0	50.1	27.5	25.3
5	119	58.4	79.7	58.7	65.1	87.9	65.1	21.0	58.4	87.9	70.0	13.2
5	122+131+142	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	124+147	0.0	0.0	0.0	0.0	17.9	0.0	0.0	0.0	17.9	3.6	8.0
5	125	55.2	92.3	57.8	58.5	53.2	57.8	3.3	53.2	92.3	63.4	16.3
6	128	49.3	158.2	74.3	76.1	246.7	76.1	83.9	49.3	246.7	120.9	81.4
6	130	78.4	275.3	94.6	135.3	120.7	120.7	40.8	78.4	275.3	140.9	78.4
6	132	42.2	97.7	43.9	52.2	55.8	52.2	11.9	42.2	97.7	58.4	22.7
6	136	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	138+163+164	47.7	89.6	48.1	57.7	59.0	57.7	11.0	47.7	89.6	60.4	17.1
6	146	119.9	477.0	165.7	259.0	219.9	219.9	93.2	119.9	477.0	248.3	138.3
6	149	132.4	314.0	192.3	205.5	218.5	205.5	26.2	132.4	314.0	212.5	65.6
6	151	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	153	78.9	117.9	76.8	77.0	95.1	78.9	18.0	76.8	117.9	89.1	17.8
6	154	54.8	195.8	60.8	88.6	81.2	81.2	27.8	54.8	195.8	96.2	57.4
6	157+201	69.9	202.2	107.4	138.7	0.0	107.4	68.8	0.0	202.2	103.6	75.6
6	158	0.0	95.2	0.0	71.5	0.0	0.0	71.5	0.0	95.2	33.3	46.4
6	183	66.7	225.5	152.2	177.1	395.5	177.1	73.4	66.7	395.5	203.4	121.8
6	165	47.8	88.9	47.7	52.4	57.8	52.4	10.0	47.7	88.9	58.9	17.2
7	170+190	65.4	176.0	45.4	66.3	57.2	65.4	9.1	45.4	176.0	82.0	53.2
7	172	0.0	71.1	71.7	0.0	145.9	71.1	71.7	0.0	145.9	57.7	60.8
7	173	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	174	61.8	244.9	78.1	54.5	143.3	78.1	81.5	54.5	244.9	116.5	79.8
7	175	60.9	180.7	208.0	170.2	410.3	180.7	37.9	60.9	410.3	206.0	127.2
7	177	0.0	89.2	0.0	0.0	0.0	0.0	0.0	0.0	89.2	17.8	39.9
7	178	0.0	0.0	39.8	0.0	52.4	0.0	39.8	0.0	52.4	18.5	25.7
7	179	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	180	0.0	110.2	60.2	53.7	0.0	53.7	60.2	0.0	110.2	44.8	46.4
7	187	45.9	144.9	107.0	130.0	203.3	130.0	37.9	45.9	203.3	126.2	57.3
7	191	57.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	57.3	11.5	25.6
7	193	0.0	154.0	67.6	57.5	107.3	67.6	49.8	0.0	154.0	77.3	57.5
8	194	48.7	92.6	149.3	0.0	64.2	64.2	43.8	0.0	149.3	71.0	55.2
8	195+208	0.0	118.8	0.0	0.0	0.0	0.0	0.0	0.0	118.8	23.8	53.1
8	196+203	0.0	82.4	60.4	74.1	78.3	74.1	14.9	0.0	82.4	58.4	33.6
8	197	104.9	311.3	115.8	188.1	157.5	157.5	72.3	104.9	311.3	175.5	82.9
8	199	49.2	118.5	91.2	62.1	142.0	91.2	56.4	49.2	142.0	92.6	38.5
8	200	42.3	103.2	39.5	49.8	0.0	42.3	10.3	0.0	103.2	46.9	36.9
8	205	46.5	91.4	0.0	0.0	0.0	0.0	46.5	0.0	91.4	27.6	41.0
9	207	47.8	156.9	54.8	61.2	0.0	54.8	13.4	0.0	156.9	64.1	57.2
Sum	ng/g lipid weight	2574.1	6580.5	3836.5	4401.3	4825.7	4401.3	989.1	2574.1	6580.5	4443.6	1464.4

5. PCB concentrations (dry weights) of Steller sea lion kidney tissues

ng/g dry weight

IUPAC No.	Tatitlek SSL-2 (1) Kidney	Tatitlek SSL-3 Kidney	Tatitlek SSL-4 Kidney	Tatitlek SSL-1 Kidney	Tatitlek SSL-2 (5) Kidney	Median	Interquartile Range	Range Min	Range Max	Average	StDev
6	0.0	4.3	0.0	0.0	0.0	0.0	0.0	0.0	4.3	0.9	1.9
16+32	0.0	0.0	0.0	0.0	3.1	0.0	0.0	0.0	3.1	0.6	1.4
22	0.0	0.0	1.3	0.0	0.0	0.0	0.0	0.0	1.3	0.3	0.6
34	0.0	0.0	0.0	8.5	0.0	0.0	0.0	0.0	8.5	1.7	3.8
26	0.0	0.0	0.0	1.8	0.0	0.0	0.0	0.0	1.8	0.4	0.8
28	0.0	3.3	0.0	0.0	0.0	0.0	0.0	0.0	3.3	0.7	1.5
35	0.0	0.0	1.7	0.0	0.0	0.0	0.0	0.0	1.7	0.3	0.8
52+73	0.0	1.6	0.0	1.8	0.0	0.0	1.6	0.0	1.8	0.7	0.9
53	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
66+93+95	0.0	1.4	0.0	1.8	0.0	0.0	1.4	0.0	1.8	0.6	0.9
74	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0	1.7	0.3	0.8
82	0.0	7.5	0.0	2.0	0.0	0.0	2.0	0.0	7.5	1.9	3.2
84	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
90+101	6.3	10.1	0.0	8.4	0.0	6.3	8.4	0.0	10.1	4.9	4.7
105	0.0	3.5	0.0	1.9	0.0	0.0	1.9	0.0	3.5	1.1	1.6
115+116+117	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
118	7.3	0.0	1.9	7.5	0.0	1.9	7.3	0.0	7.5	3.3	3.8
119	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.1	0.4	0.9
122+131+142	0.0	20.0	0.0	0.0	0.0	0.0	0.0	0.0	20.0	4.0	9.0
128	0.0	2.7	0.0	2.3	1.8	1.8	2.3	0.0	2.7	1.4	1.3
130	0.0	1.9	0.0	0.0	0.0	0.0	0.0	0.0	1.9	0.4	0.8
138+163+164	3.8	5.9	1.6	5.6	0.0	3.8	4.0	0.0	5.9	3.4	2.6
146	2.3	0.0	0.0	2.7	0.0	0.0	2.3	0.0	2.7	1.0	1.4
149	2.2	3.3	0.0	0.0	0.0	0.0	2.2	0.0	3.3	1.1	1.6
151	0.0	1.9	0.0	0.0	0.0	0.0	0.0	0.0	1.9	0.4	0.8
153	9.3	8.4	1.6	16.8	0.0	8.4	7.7	0.0	16.8	7.2	6.7
154	0.0	15.3	0.0	4.0	0.0	0.0	4.0	0.0	15.3	3.9	6.6
162+183	1.5	1.5	0.0	3.3	0.0	1.5	1.5	0.0	3.3	1.3	1.4
170+190	0.0	3.7	0.0	3.6	1.7	1.7	3.6	0.0	3.7	1.8	1.8
171+202	0.0	9.5	0.0	0.0	0.0	0.0	0.0	0.0	9.5	1.9	4.2
175	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0	2.4	0.5	1.1
177	0.0	3.8	0.0	0.0	1.9	0.0	1.9	0.0	3.8	1.1	1.7
178	0.0	0.0	0.0	1.6	0.0	0.0	0.0	0.0	1.6	0.3	0.7
180	5.2	8.8	0.0	7.8	0.0	5.2	7.8	0.0	8.8	4.4	4.2
187	0.0	2.3	0.0	1.7	0.0	0.0	1.7	0.0	2.3	0.8	1.1
194	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0	1.7	0.3	0.7
197	0.0	0.0	1.9	0.0	0.0	0.0	0.0	0.0	1.9	0.4	0.9
199	0.0	2.6	0.0	0.0	0.0	0.0	0.0	0.0	2.6	0.5	1.2
200	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ng/g dry weight	40.0	125.4	10.0	86.4	8.5	40.0	76.4	8.5	125.4	54.0	50.9

ng/g dry weight

Location Sample ID Cl-No.	IUPAC No.	St. Paul SNPSLS9901 Kidney	St. Paul SNPSLS9802 Kidney	St. Paul SNPSLS2000-04 Kidney	St. Paul SNPSLS9808 Kidney	Median	Interquartile Range	Range Min	Range Max	Average	StDev
2	6	0.0	6.8	4.3	3.7	4.0	2.1	0.0	6.8	3.7	2.8
3	16+32	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	22	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	34	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	26	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	28	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	35	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	52+73	1.5	0.0	0.0	0.0	0.0	0.4	0.0	1.5	0.4	0.8
4	53	0.0	0.0	5.6	5.2	2.6	5.3	0.0	5.6	2.7	3.1
4	66+93+95	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	74	0.0	0.0	3.7	0.0	0.0	0.9	0.0	3.7	0.9	1.9
5	82	2.3	2.4	0.0	0.0	1.2	2.3	0.0	2.4	1.2	1.4
5	84	2.3	0.0	2.0	0.0	1.0	2.1	0.0	2.3	1.1	1.2
5	90+101	3.7	9.6	4.3	6.3	5.3	3.0	3.7	9.6	6.0	2.6
5	105	3.0	0.0	0.0	0.0	0.0	0.7	0.0	3.0	0.7	1.5
5	115+116+117	0.0	0.0	32.3	20.4	10.2	23.4	0.0	32.3	13.2	16.0
5	118	3.9	5.6	3.5	3.2	3.7	0.9	3.2	5.6	4.0	1.1
5	119	0.0	0.0	0.0	2.0	0.0	0.5	0.0	2.0	0.5	1.0
5	122+131+142	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	128	2.0	2.1	1.6	0.0	1.8	0.9	0.0	2.1	1.4	1.0
6	130	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	138+163+164	2.6	4.4	2.2	2.9	2.8	0.7	2.2	4.4	3.0	1.0
6	146	1.4	0.0	0.0	0.0	0.0	0.4	0.0	1.4	0.4	0.7
6	149	1.3	1.5	0.0	0.0	0.7	1.3	0.0	1.5	0.7	0.8
6	151	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	153	4.9	11.5	4.5	4.5	4.8	1.8	4.5	11.5	6.4	3.4
6	154	5.3	6.2	2.0	2.5	3.9	3.2	2.0	6.2	4.0	2.1
6	162+183	1.7	2.7	0.0	1.8	1.7	0.8	0.0	2.7	1.5	1.1
7	170+190	2.1	4.7	1.6	1.8	1.9	1.0	1.6	4.7	2.5	1.5
7	171+202	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	175	2.2	0.0	0.0	0.0	0.0	0.6	0.0	2.2	0.6	1.1
7	177	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	178	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	180	0.0	7.4	3.0	3.6	3.3	2.3	0.0	7.4	3.5	3.0
7	187	0.0	2.0	0.0	0.0	0.0	0.5	0.0	2.0	0.5	1.0
8	194	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8	197	4.6	0.0	0.0	0.0	0.0	1.2	0.0	4.6	1.2	2.3
8	199	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8	200	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sum	ng/g dry weight	44.8	66.8	70.7	57.7	62.3	13.4	44.8	70.7	60.0	11.5

6. PCB concentrations (lipid weights) of Steller sea lion kidney tissues

Location Sample ID Cl-No.	ng/g lipid weight IUPAC No.	Tatitlek SSL-2 (1) Kidney	Tatitlek SSL-3 Kidney	Tatitlek SSL-4 Kidney	Tatitlek SSL-1 Kidney	Tatitlek SSL-2 (5) Kidney	Median	Interquartile Range	Range Min	Range Max	Average	StDev
2	6	0.0	76.7	0.0	0.0	0.0	0.0	0.0	0.0	76.7	15.3	34.3
3	16+32	0.0	0.0	0.0	0.0	136.1	0.0	0.0	0.0	136.1	27.2	60.9
3	22	0.0	0.0	17.1	0.0	0.0	0.0	0.0	0.0	17.1	3.4	7.7
3	34	0.0	0.0	0.0	157.7	0.0	0.0	0.0	0.0	157.7	31.5	70.5
3	26	0.0	0.0	0.0	33.5	0.0	0.0	0.0	0.0	33.5	6.7	15.0
3	28	0.0	58.2	0.0	0.0	0.0	0.0	0.0	0.0	58.2	11.6	26.0
3	35	0.0	0.0	23.4	0.0	0.0	0.0	0.0	0.0	23.4	4.7	10.5
4	52+73	0.0	27.6	0.0	32.4	0.0	0.0	27.6	0.0	32.4	12.0	16.5
4	53	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	66+93+95	0.0	24.6	0.0	33.9	0.0	0.0	24.6	0.0	33.9	11.7	16.4
4	74	0.0	0.0	0.0	31.3	0.0	0.0	0.0	0.0	31.3	6.3	14.0
5	82	0.0	133.0	0.0	37.8	0.0	0.0	37.8	0.0	133.0	34.2	57.6
5	84	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	90+101	75.4	179.1	0.0	154.9	0.0	75.4	154.9	0.0	179.1	81.9	84.0
5	105	0.0	61.8	0.0	35.9	0.0	0.0	35.9	0.0	61.8	19.5	28.3
5	115+116+117	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5	118	87.7	0.0	25.9	139.5	0.0	25.9	87.7	0.0	139.5	50.6	61.3
5	119	24.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	24.9	5.0	11.1
5	122+131+142	0.0	356.4	0.0	0.0	0.0	0.0	0.0	0.0	356.4	71.3	159.4
6	128	0.0	47.4	0.0	43.2	76.7	43.2	47.4	0.0	76.7	33.5	33.2
6	130	0.0	33.8	0.0	0.0	0.0	0.0	0.0	0.0	33.8	6.8	15.1
6	138+163+164	46.2	105.2	21.9	103.6	0.0	46.2	81.7	0.0	105.2	55.4	47.6
6	146	27.5	0.0	0.0	49.1	0.0	0.0	27.5	0.0	49.1	15.3	22.3
6	149	26.8	59.1	0.0	0.0	0.0	0.0	26.8	0.0	59.1	17.2	26.2
6	151	0.0	33.6	0.0	0.0	0.0	0.0	0.0	0.0	33.6	6.7	15.0
6	153	111.6	149.7	22.1	311.1	0.0	111.6	127.7	0.0	311.1	118.9	124.0
6	154	0.0	271.9	0.0	73.9	0.0	0.0	73.9	0.0	271.9	69.2	117.8
6	162+183	18.3	27.1	0.0	60.6	0.0	18.3	27.1	0.0	60.6	21.2	25.0
7	170+190	0.0	65.3	0.0	65.8	73.7	65.3	65.8	0.0	73.7	41.0	37.5
7	171+202	0.0	169.0	0.0	0.0	0.0	0.0	0.0	0.0	169.0	33.8	75.6
7	175	0.0	41.8	0.0	0.0	0.0	0.0	0.0	0.0	41.8	8.4	18.7
7	177	0.0	67.8	0.0	0.0	81.5	0.0	67.8	0.0	81.5	29.9	41.2
7	178	0.0	0.0	0.0	29.1	0.0	0.0	0.0	0.0	29.1	5.8	13.0
7	180	62.2	156.1	0.0	144.9	0.0	62.2	144.9	0.0	156.1	72.6	75.6
7	187	0.0	40.6	0.0	31.7	0.0	0.0	31.7	0.0	40.6	14.5	20.0
8	194	0.0	0.0	0.0	30.9	0.0	0.0	0.0	0.0	30.9	6.2	13.8
8	197	0.0	0.0	26.6	0.0	0.0	0.0	0.0	0.0	26.6	5.3	11.9
8	199	0.0	46.5	0.0	0.0	0.0	0.0	0.0	0.0	46.5	9.3	20.8
8	200	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sum	ng/g lipid weight	480.5	2232.5	137.0	1601.0	368.1	480.5	1232.9	137.0	2232.5	963.8	906.6

Location Sample ID Cl-No.	ng/g lipid weight IUPAC No.	St. Paul SLS9901 Kidney	St. Paul SSL9802SNP Kidney	St. Paul SNPSLS2000-04 Kidney	St. Paul SLS9803 Kidney	Median	Interquartile Range	Range Min	Range Max	Average	StDev
2	6	0.0	166.9	75.4	90.3	82.8	52.9	0.0	166.9	83.1	68.4
3	16+32	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	22	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	34	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	26	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	28	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3	35	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	52+73	38.2	0.0	0.0	0.0	0.0	9.5	0.0	38.2	9.5	19.1
4	53	0.0	0.0	99.3	124.5	49.7	105.6	0.0	124.5	56.0	65.4
4	66+93+95	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4	74	0.0	0.0	66.4	0.0	0.0	16.6	0.0	66.4	16.6	33.2
5	82	58.0	59.7	0.0	0.0	29.0	58.5	0.0	59.7	29.4	34.0
5	84	56.8	0.0	35.3	0.0	17.7	40.7	0.0	56.8	23.0	28.0
5	90+101	93.0	233.9	76.1	151.6	122.3	83.4	76.1	233.9	138.6	71.3
5	105	74.6	0.0	0.0	0.0	0.0	18.7	0.0	74.6	18.7	37.3
5	115+116+117	0.0	0.0	573.0	491.6	245.8	512.0	0.0	573.0	266.2	309.1
5	118	97.7	137.3	61.2	76.5	87.1	34.9	61.2	137.3	93.2	33.0
5	119	0.0	0.0	0.0	48.8	0.0	12.2	0.0	48.8	12.2	24.4
5	122+131+142	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	128	49.7	52.1	27.5	0.0	38.6	29.7	0.0	52.1	32.3	24.2
6	130	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	138+163+164	65.8	107.7	38.9	69.5	67.7	20.0	38.9	107.7	70.5	28.3
6	146	35.7	0.0	0.0	0.0	0.0	8.9	0.0	35.7	8.9	17.8
6	149	32.9	35.5	0.0	0.0	16.5	33.6	0.0	35.5	17.1	19.8
6	151	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6	153	121.9	281.6	84.6	107.4	114.6	60.1	84.6	281.6	148.9	89.8
6	154	133.9	152.2	34.6	59.6	96.8	85.1	34.6	152.2	95.1	56.8
6	162+183	42.2	64.8	0.0	43.4	42.8	17.1	0.0	64.8	37.6	27.2
7	170+190	52.5	114.5	28.0	42.5	47.5	29.1	28.0	114.5	59.4	38.1
7	171+202	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	175	55.8	0.0	0.0	0.0	0.0	13.9	0.0	55.8	13.9	27.9
7	177	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	178	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7	180	0.0	181.1	53.4	85.9	69.7	69.7	0.0	181.1	80.1	76.1
7	187	0.0	48.2	0.0	0.0	0.0	12.1	0.0	48.2	12.1	24.1
8	194	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8	197	115.6	0.0	0.0	0.0	0.0	28.9	0.0	115.6	28.9	57.8
8	199	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8	200	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sum	ng/g lipid weight	1124.4	1635.6	1253.8	1391.6	1322.7	231.2	1124.4	1635.6	1351.3	218.7

7. PCB concentrations (dry weights) of Steller sea lion placenta tissues

ng/g dry weight

Sample ID Cl-No.	IUPAC No.	Japan Placenta	Rock Placenta	Ugamak Placenta	Unalask (a) Placenta	Unalask (b) Placenta	Rock (a) Placenta	Rock (b) Placenta	Rock c Placenta	Rock (d) Placenta	Median	Interquartile Range	Range Min	Range Max	Average	StDev
3	19	0.0	2.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.6	0.3	0.9
4	41+71	0.0	0.0	6.4	0.0	6.3	0.0	0.0	6.4	0.0	0.0	6.3	0.0	6.4	2.1	3.2
4	44	0.0	0.0	0.0	0.0	2.0	0.0	2.3	0.0	0.0	0.0	0.0	0.0	2.3	0.5	0.9
4	45	3.1	0.0	0.0	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.1	0.6	1.2
4	47+48+75	0.0	0.0	0.0	0.0	0.0	2.2	0.0	0.0	0.0	0.0	0.0	0.0	2.2	0.2	0.7
4	52+73	1.2	1.2	1.5	1.8	1.8	1.9	1.3	0.0	0.0	1.3	0.6	0.0	1.9	1.2	0.7
4	56+60	0.0	0.0	0.0	0.0	1.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.3	0.1	0.4
4	66+93+95	0.0	1.1	1.8	1.7	2.2	0.0	0.0	1.6	0.0	1.1	1.7	0.0	2.2	0.9	0.9
4	74	0.0	0.0	1.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.9	0.2	0.6
4	81+87+111	0.0	0.0	1.4	0.0	3.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.1	0.5	1.1
5	82	4.0	0.0	1.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.0	0.6	1.4
5	83	3.0	1.9	2.3	0.0	2.7	2.0	2.7	1.5	0.0	2.0	1.2	0.0	3.0	1.8	1.1
5	84	1.6	1.2	2.4	2.6	4.4	1.5	1.4	2.7	2.1	2.1	1.1	1.2	4.4	2.2	1.0
5	90+101	4.0	1.4	7.7	3.0	4.1	0.0	3.1	4.0	0.0	3.1	2.6	0.0	7.7	3.0	2.4
5	105	3.5	0.0	0.0	0.0	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.5	0.5	1.2
5	118	11.5	2.3	12.0	7.7	8.4	3.6	2.3	6.4	3.8	6.4	4.8	2.3	12.0	6.4	3.7
5	119	5.3	0.0	1.3	0.0	0.0	0.0	0.0	3.4	0.0	0.0	1.3	0.0	5.3	1.1	1.9
6	128	2.5	0.0	3.6	0.0	0.0	2.3	0.0	0.0	0.0	0.0	2.3	0.0	3.6	0.9	1.4
6	136	0.0	2.3	3.6	4.6	4.1	1.5	0.0	5.0	0.0	2.3	4.1	0.0	5.0	2.3	2.1
6	138+163+164	3.2	0.0	5.6	1.9	2.2	0.0	1.2	1.9	1.3	1.9	1.0	0.0	5.6	1.9	1.7
6	149	1.2	0.0	1.2	0.0	1.4	0.0	0.0	1.2	0.0	0.0	1.2	0.0	1.4	0.6	0.7
6	153	5.4	4.9	18.0	6.0	6.7	6.2	1.7	3.9	1.7	5.4	2.3	1.7	18.0	6.1	4.8
6	154	0.0	0.0	0.0	2.5	7.4	1.6	6.6	0.0	0.0	0.0	2.5	0.0	7.4	2.0	3.0
6	162	0.0	0.0	3.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.3	0.4	1.1
7	170+190	2.4	0.0	3.2	0.0	0.0	0.0	0.0	2.3	0.0	0.0	2.3	0.0	3.2	0.9	1.3
7	171+202	1.2	0.0	0.0	0.0	0.0	0.0	3.9	0.0	0.0	0.0	0.0	0.0	3.9	0.6	1.3
7	172	0.0	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.0	0.0	1.5	0.2	0.5
7	177	5.3	0.0	0.0	0.0	3.3	0.0	0.0	0.0	12.8	0.0	3.3	0.0	12.8	2.4	4.4
7	180	5.9	0.0	7.4	1.5	2.1	0.0	0.0	0.0	0.0	0.0	2.1	0.0	7.4	1.9	2.8
7	187	2.3	0.0	1.3	0.0	0.0	1.8	0.0	0.0	0.0	0.0	1.3	0.0	2.3	0.6	0.9
7	191	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0	2.0	0.2	0.7
8	194	2.6	0.0	1.8	2.4	0.0	0.0	3.1	2.7	0.0	1.8	2.6	0.0	3.1	1.4	1.4
8	196+203	1.3	0.0	1.1	1.1	1.9	0.0	0.0	0.0	0.0	0.0	1.1	0.0	1.9	0.6	0.8
8	199	0.0	0.0	1.4	1.8	0.0	0.0	11.1	0.0	1.4	0.0	1.4	0.0	11.1	1.7	3.6
8	200	0.0	0.0	0.0	1.8	0.0	5.1	0.0	0.0	1.6	0.0	1.6	0.0	5.1	0.9	1.7
9	206	0.0	0.0	4.8	0.0	0.0	0.0	0.0	6.6	0.0	0.0	0.0	0.0	6.6	1.3	2.6
Sum	ng/g dry weight	70.3	19.0	96.9	40.4	68.9	29.7	44.0	49.7	24.7	44.0	39.3	19.0	96.9	49.3	25.3

ng/g lipid weight

Sample ID Cl-No.	IUPAC No.	Japan Placenta	Rock Placenta	Ugamak Placenta	Unalask (a) Placenta	Unalask (b) Placenta	Rock (a) Placenta	Rock (b) Placenta	Rock c Placenta	Rock (d) Placenta	Median	Interquartile Range	Range Min	Range Max	Average	StDev
3	19	0.0	57.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	57.4	6.4	19.1
4	41+71	0.0	0.0	124.0	0.0	93.2	0.0	0.0	87.5	0.0	0.0	87.5	0.0	124.0	33.9	51.7
4	44	0.0	0.0	0.0	0.0	29.3	0.0	343.5	0.0	0.0	0.0	0.0	0.0	343.5	41.4	113.7
4	45	51.6	0.0	0.0	0.0	36.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	51.6	9.7	19.7
4	47+48+75	0.0	0.0	0.0	0.0	0.0	57.9	0.0	0.0	0.0	0.0	0.0	0.0	57.9	6.4	19.3
4	52+73	19.6	25.6	29.7	60.1	27.3	49.2	189.0	0.0	0.0	27.3	29.7	0.0	189.0	44.5	57.7
4	56+60	0.0	0.0	0.0	0.0	18.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	18.5	2.1	6.2
4	66+93+95	0.0	23.9	34.0	58.1	32.9	0.0	0.0	22.2	0.0	22.2	32.9	0.0	58.1	19.0	20.7
4	74	0.0	0.0	36.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	36.9	4.1	12.3
4	81+87+111	0.0	0.0	27.2	0.0	45.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	45.5	8.1	16.7
5	82	67.1	0.0	34.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	67.1	11.3	23.9
5	83	50.4	41.2	45.4	0.0	40.0	52.4	403.5	20.0	0.0	41.2	30.4	0.0	403.5	72.5	125.7
5	84	26.3	26.7	47.4	88.3	64.6	40.1	205.5	36.8	115.0	47.4	51.6	26.3	205.5	72.3	58.1
5	90+101	68.0	31.1	149.8	101.2	61.2	0.0	463.5	55.1	0.0	61.2	70.1	0.0	463.5	103.3	143.1
5	105	58.9	0.0	0.0	0.0	17.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	58.9	8.4	19.7
5	118	194.1	51.3	231.9	261.5	124.1	94.0	343.5	88.2	208.6	194.1	138.0	51.3	343.5	177.5	95.2
5	119	89.2	0.0	26.0	0.0	0.0	0.0	0.0	47.2	0.0	0.0	26.0	0.0	89.2	18.0	31.5
6	128	42.3	0.0	69.7	0.0	0.0	59.4	0.0	0.0	0.0	0.0	42.3	0.0	69.7	19.0	29.4
6	136	0.0	50.2	69.1	157.6	61.2	38.0	0.0	68.7	0.0	50.2	68.7	0.0	157.6	49.4	50.1
6	138+163+164	54.0	0.0	109.1	62.8	32.8	0.0	184.5	26.3	72.8	54.0	46.5	0.0	184.5	60.3	58.3
6	149	20.7	0.0	23.1	0.0	20.2	0.0	0.0	16.5	0.0	0.0	20.2	0.0	23.1	8.9	10.7
6	153	90.9	107.8	350.1	204.1	99.8	162.6	253.5	53.9	94.7	107.8	109.4	53.9	350.1	157.5	95.8
6	154	0.0	0.0	0.0	85.9	109.7	42.1	988.5	0.0	0.0	0.0	85.9	0.0	988.5	136.3	322.3
6	162	0.0	0.0	64.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	64.1	7.1	21.4
7	170+190	40.3	0.0	62.5	0.0	0.0	0.0	0.0	31.5	0.0	0.0	31.5	0.0	62.5	14.9	23.8
7	171+202	20.1	0.0	0.0	0.0	0.0	0.0	588.0	0.0	0.0	0.0	0.0	0.0	588.0	67.6	195.3
7	172	0.0	0.0	0.0	0.0	0.0	0.0	217.5	0.0	0.0	0.0	0.0	0.0	217.5	24.2	72.5
7	177	88.5	0.0	0.0	0.0	49.4	0.0	0.0	0.0	700.7	0.0	49.4	0.0	700.7	93.2	230.0
7	180	99.2	0.0	144.0	49.9	30.4	0.0	0.0	0.0	0.0	0.0	49.9	0.0	144.0	35.9	52.9
7	187	38.1	0.0	25.0	0.0	0.0	48.2	0.0	0.0	0.0	0.0	25.0	0.0	48.2	12.4	19.4
7	191	0.0	0.0	0.0	0.0	0.0	0.0	294.0	0.0	0.0	0.0	0.0	0.0	294.0	32.7	98.0
8	194	43.2	0.0	34.2	81.5	0.0	0.0	460.5	37.2	0.0	34.2	43.2	0.0	460.5	72.9	148.0
8	196+203	22.4	0.0	21.0	37.4	28.8	0.0	0.0	0.0	0.0	0.0	22.4	0.0	37.4	12.2	15.1
8	199	0.0	0.0	27.2	61.8	0.0	0.0	1665.0	0.0	73.9	0.0	61.8	0.0	1665.0	203.1	549.0
8	200	0.0	0.0	0.0	60.1	0.0	132.7	0.0	0.0	87.6	0.0	60.1	0.0	132.7	31.2	50.2
9	206	0.0	0.0	93.7	0.0	0.0	0.0	0.0	90.8	0.0	0.0	0.0	0.0	93.7	20.5	40.7
Sum	ng/g v lipid weight	1185.0	415.2	1879.8	1370.4	1021.8	776.5	6600.0	681.8	1353.3	1185.0	593.9	415.2	6600.0	1698.2	1888.7

7. Lipid weights (%) of all Steller sea lion tissues (blubber, liver, kidney and placenta)

Tissue type	Bluber			Blubber					
Location	Tatitlek	Tatitlek	Tatitlek	St. Paul	St. Paul	St. Paul	St. Paul	St. Paul	St. Paul
Sample ID	SSL-3	SSL-4	SSL-2 (5)	SNPSLS9802	SNPSLS9901	SNPSLS2000-04	SP-01-00-EJ	SNPSLS9803	Karin (SP-01-01-EJ)
% Lipid	42	52	37	52	55	31	52	46	43

Tissue type	Liver					Liver				
Location	Tatitlek	Tatitlek	Tatitlek	Tatitlek	Tatitlek	St. Paul	St. Paul	St. Paul	St. Paul	St. Paul
Sample ID	SSL-2 (1)	SSL-3	SSL-4	SSL-1	SSL-2 (5)	SNPSLS9901	SSL9802SNP	SNPSLS2000-04	SNPSLS9808	Karin (SP-01-01-EJ)
% Lipid	9.1	6.1	6.3	5.8	3.8	5.6	3.3	6.0	5.1	4.9

Tissue type	Kidney					Kidney			
Location	Tatitlek	Tatitlek	Tatitlek	Tatitlek	Tatitlek	St. Paul	St. Paul	St. Paul	St. Paul
Sample ID	SSL-2 (1)	SSL-3	SSL-4	SSL-1	SSL-2 (5)	SNPSLS9901	SSL9802SNP	SNPSLS2000-04	SNPSLS9808
% Lipid	8.3	5.6	7.3	5.4	2.3	4.0	4.1	5.6	4.1

Tissue type	Placenta								
Location	Japan	Rock	Ugamak	Unalask (a)	Unalask (b)	Rock (a)	Rock (b)	Rock c	Rock (d)
% Lipid	5.9	4.6	5.2	2.9	6.7	3.8	0.7	7.3	1.8

Appendix C: PCB and OCP original data for Steller sea lion blubber and liver from chapter 3

1. PCB concentration (dry weights) of Steller sea lion blubber samples (all locations)

ng/g dry weight

Location ID Congeners	Olutorsky Gulf (Russia) 1-Ageev Blubber	2-Ageev Blubber	3-Ageev Blubber	1-Testin Blubber	2-Testin Blubber	3-Testin Blubber	1-Udalov Blubber	2- Udalov Blubber	3-Udalov Blubber	1-Vozikov Blubber	Median	Interquartile Range	Range Min	Range Max
PCB 8	0.7	0.9	0.6	0.7	1.0	0.9	1.1	0.7	0.7	1.0	0.8	0.3	0.6	1.1
PCB 18	13.9	7.9	2.0	2.7	10.8	4.1	7.8	4.7	2.5	5.0	4.8	4.8	2.0	10.8
PCB 28	23.4	16.8	14.1	14.8	19.5	6.8	13.6	4.0	11.3	16.6	14.4	4.9	4.0	19.5
PCB 52	223.4	95.5	32.5	26.8	100.1	43.1	109.6	77.0	26.6	37.5	60.1	65.2	26.6	109.6
PCB 44	56.4	29.9	7.2	6.3	32.4	13.6	35.1	20.8	5.4	10.0	17.2	23.9	5.4	35.1
PCB 66	77.1	36.4	28.1	28.2	41.6	19.2	38.2	11.9	33.3	28.4	30.9	9.6	11.9	41.6
PCB 81	nd	nd	nd	nd	nd	nd	0.1	nd	nd	nd	0.1	0.0	0.1	0.1
PCB 77	1.4	2.0	nd	nd	0.8	0.4	1.5	1.0	nd	136.4	1.4	0.9	0.4	136.4
PCB 101	263.5	110.9	13.0	16.4	95.2	57.9	125.4	96.4	10.7	19.3	76.6	90.1	10.7	125.4
PCB 123	266.4	145.6	5.7	nd	139.0	96.4	91.2	110.2	139.6	nd	124.6	46.0	5.7	145.6
PCB 118	257.5	140.4	132.7	109.5	132.8	94.5	123.3	104.0	133.5	109.6	128.0	23.8	94.5	140.4
PCB 114	8.9	2.2	2.8	1.3	3.6	1.8	3.2	2.0	4.1	3.1	3.0	1.4	1.3	4.1
PCB 105	84.0	37.8	35.1	35.3	39.6	20.6	42.8	28.3	37.6	36.7	37.2	4.0	20.6	42.8
PCB 153	653.6	278.2	313.6	240.3	274.1	160.8	261.4	232.9	286.2	187.7	267.7	49.4	160.8	313.6
PCB 138	414.3	191.3	143.5	102.1	178.7	109.8	163.6	159.9	176.6	112.8	161.7	57.7	102.1	191.3
PCB 126	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 128	63.5	30.5	33.4	27.9	27.3	16.1	30.4	24.1	27.2	23.2	27.6	5.6	16.1	33.4
PCB 167	3.8	1.5	1.8	2.1	2.1	1.3	1.5	nd	2.8	1.7	1.8	0.6	1.3	2.8
PCB 156	4.6	1.0	2.4	1.8	2.8	1.2	1.4	nd	3.5	1.6	1.8	1.4	1.0	3.5
PCB 157	6.3	2.0	0.2	2.0	3.0	1.5	2.5	0.8	3.4	0.8	2.0	1.9	0.2	3.4
PCB 169	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 187	86.8	30.4	7.8	7.3	22.3	13.6	35.5	34.9	5.8	7.6	17.9	26.1	5.8	35.5
PCB 180	134.6	61.9	67.1	49.4	57.4	36.9	64.5	62.1	62.8	38.2	62.0	12.7	36.9	67.1
PCB 170	46.9	19.1	19.1	15.0	17.6	11.5	19.4	17.9	12.1	7.8	17.7	6.3	7.8	19.4
PCB 189	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 195	2.7	1.0	2.5	2.2	1.1	0.6	1.3	1.0	1.1	1.4	1.2	0.9	0.6	2.5
PCB 206	2.4	1.0	2.7	1.3	1.2	0.5	1.3	1.3	1.1	1.3	1.3	0.2	0.5	2.7
PCB 209	1.5	0.9	1.7	1.2	0.9	0.5	1.1	1.1	0.8	0.9	1.0	0.3	0.5	1.7
Sum PCB	2697.5	1245.0	869.8	694.4	1204.8	713.6	1176.5	997.1	988.5	788.6	992.8	388.8	694.4	1245.0

Location ID Congeners	ng/g dry weight St. Paul SSLSNP 2001-03 Blubber	SNPSLS 9804 Blubber	SNPSLS 9801 Blubber	SSLSNP 2001-05 Blubber	SNPSLS9803 Blubber	SNPSLS9902 Blubber	Median	Interquartile Range	Range Min	Range Max
PCB 8	2.4	1.6	1.6	0.9	1.5	2.6	1.6	0.7	0.9	2.6
PCB 18	3.9	4.0	3.5	1.8	2.5	6.1	3.5	1.6	1.8	6.1
PCB 28	11.2	12.6	19.2	7.3	21.6	30.4	19.2	11.0	7.3	30.4
PCB 52	26.2	64.1	51.9	13.8	46.2	44.0	46.2	21.8	13.8	64.1
PCB 44	5.9	11.0	4.8	2.1	7.6	8.7	6.1	2.8	2.1	11.0
PCB 66	62.9	64.3	80.8	114.5	61.3	71.1	71.1	18.3	61.3	114.5
PCB 81	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 77	0.4	nd	nd	nd	1.6	nd	1.0	0.6	0.4	1.6
PCB 101	15.3	53.5	28.7	15.7	21.5	23.5	23.5	11.4	15.3	53.5
PCB 123	2.9	nd	nd	nd	nd	84.3	43.6	40.7	2.9	84.3
PCB 118	316.9	213.5	261.3	503.1	188.1	286.2	261.3	71.7	188.1	503.1
PCB 114	7.3	5.7	6.9	16.9	0.0	6.4	6.4	1.1	0.0	16.9
PCB 105	91.7	64.0	88.2	165.4	62.8	112.9	91.7	28.7	62.8	165.4
PCB 153	725.6	442.1	551.9	1041.6	340.4	826.3	584.7	279.0	340.4	1041.6
PCB 138	384.3	290.0	318.5	691.6	196.2	419.5	318.5	119.3	196.2	691.6
PCB 126	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 128	83.6	57.0	67.7	127.0	49.1	102.0	74.1	30.4	49.1	127.0
PCB 167	8.5	5.2	6.6	8.6	5.5	5.0	6.6	3.1	5.0	8.6
PCB 156	12.7	5.3	8.4	6.4	5.0	4.2	6.4	4.5	4.2	12.7
PCB 157	11.1	nd	nd	7.7	0.8	1.1	2.2	6.5	0.8	11.1
PCB 169	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 187	17.9	32.9	23.2	23.3	14.7	16.1	22.5	6.2	14.7	32.9
PCB 180	151.8	127.8	130.9	179.7	73.8	156.7	130.9	34.6	73.8	179.7
PCB 170	53.3	44.2	41.8	61.3	24.0	57.2	44.2	14.6	24.0	61.3
PCB 189	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 195	3.5	2.6	2.2	2.5	2.5	6.4	2.6	1.6	2.2	6.4
PCB 206	4.1	2.6	3.4	1.8	2.0	4.0	2.8	1.4	1.8	4.1
PCB 209	1.7	1.0	1.2	1.0	0.9	1.7	1.2	0.4	0.9	1.7
Sum PCB	2005.2	1505.3	1702.6	2994.2	1129.6	2276.3	1702.6	539.9	1129.6	2994.2

Location ID Congeners	ng/g dry weight Tatitlek (males) Tatitlek # 4 Blubber	Tatitlek # 9 Blubber	Tatitlek # 3 Blubber	Tatitlek # 1 Blubber	Tatitlek 10 Blubber	Tatitlek 7 Blubber	Tatitlek 8 Blubber	Median	Interquartile Range	Range Min	Range Max
PCB 8	2.1	4.1	0.4	nd	1.2	1.8	1.5	1.7	0.8	0.4	4.1
PCB 18	5.6	11.1	0.7	2.9	5.7	4.3	4.2	4.3	2.1	0.7	11.1
PCB 28	32.1	12.3	6.6	9.6	21.3	13.9	15.1	13.9	7.3	6.6	32.1
PCB 52	60.4	112.9	20.8	40.3	34.1	26.1	39.2	39.2	20.3	20.8	112.9
PCB 44	7.9	22.8	2.1	7.6	9.4	7.4	7.9	7.9	1.2	2.1	22.8
PCB 66	28.7	22.4	27.8	26.9	12.4	14.2	16.0	22.4	12.3	12.4	28.7
PCB 81	0.4	0.6	nd	nd	nd	nd	nd	0.5	0.1	0.4	0.6
PCB 77	nd	nd	nd	nd	0.5	nd	nd	0.5	0.0	0.5	0.5
PCB 101	51.2	144.7	25.8	38.5	22.1	15.8	22.3	25.8	22.7	15.8	144.7
PCB 123	nd	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 118	95.4	128.0	108.3	113.5	39.7	29.7	47.3	95.4	67.4	29.7	128.0
PCB 114	3.0	5.5	3.7	4.1	1.3	0.8	0.6	3.0	2.8	0.6	5.5
PCB 105	20.4	38.9	36.0	35.9	12.3	8.4	6.7	20.4	25.6	6.7	38.9
PCB 153	133.4	451.3	227.4	341.4	94.8	62.0	116.5	133.4	178.7	62.0	451.3
PCB 138	98.7	253.4	154.1	204.2	56.8	41.0	63.1	98.7	119.2	41.0	253.4
PCB 126	nd	nd	ndn	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 128	13.0	39.1	22.7	28.4	8.8	6.4	10.3	13.0	16.0	6.4	39.1
PCB 167	2.1	1.5	3.7	3.2	nd	0.5	0.8	1.8	1.9	0.5	3.7
PCB 156	0.6	0.7	3.6	3.0	0.8	nd	nd	0.8	2.3	0.6	3.6
PCB 157	nd	0.8	2.0	3.1	0.6	0.1	1.4	1.1	1.2	0.1	3.1
PCB 169	nd	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 187	16.4	59.9	26.4	33.6	11.7	6.4	11.9	16.4	18.2	6.4	59.9
PCB 180	29.7	98.0	68.9	138.3	22.9	13.9	23.6	29.7	60.2	13.9	138.3
PCB 170	10.1	29.2	21.7	46.7	6.4	4.7	6.9	10.1	18.7	4.7	46.7
PCB 189	nd	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 195	0.4	1.5	2.2	4.1	1.3	0.6	0.4	1.3	1.4	0.4	4.1
PCB 206	nd	2.3	3.2	2.4	nd	nd	0.8	2.4	0.6	0.8	3.2
PCB 209	nd	1.4	1.6	1.0	0.2	0.3	nd	1.0	1.1	0.2	1.6
Sum PCB	611.8	1442.3	769.7	1088.7	364.4	258.3	396.7	611.8	548.6	258.3	1442.3

Location ID Congeners	ng/g dry weight Tatitlek (females) Tatitlek 5 Blubber	Tatitlek 6 Blubber	Tatitlek 3 Blubber	Tatitlek 1 Blubber	Tatitlek 2 Blubber	Median	Interquartile Range	Range Min	Max
PCB 8	2.9	5.4	0.2	3.1	1.3	2.9	1.8	0.2	5.4
PCB 18	4.6	9.3	0.3	5.1	1.5	4.6	3.6	0.3	9.3
PCB 28	13.4	30.1	1.1	40.3	8.9	13.4	21.2	1.1	40.3
PCB 52	29.3	28.7	3.2	55.5	10.1	28.7	19.2	3.2	55.5
PCB 44	9.4	14.0	0.5	12.2	5.1	9.4	7.1	0.5	14.0
PCB 66	28.6	18.0	4.0	76.1	3.3	18.0	24.6	3.3	76.1
PCB 81	0.2	nd	nd	nd	nd	0.2	0.0	0.2	0.2
PCB 77	34.7	0.9	nd	nd	nd	17.8	16.9	0.9	34.7
PCB 101	30.5	29.7	4.7	39.1	9.0	29.7	21.5	4.7	39.1
PCB 123	nd	nd	nd	38.4	nd	38.4	0.0	38.4	38.4
PCB 118	87.4	34.6	20.5	220.5	5.3	34.6	66.9	5.3	220.5
PCB 114	2.0	1.0	0.4	5.8	0.1	1.0	1.6	0.1	5.8
PCB 105	27.9	8.5	5.8	89.7	1.1	8.5	22.2	1.1	89.7
PCB 153	192.7	63.6	39.1	661.3	16.0	63.6	153.7	16.0	661.3
PCB 138	97.6	36.2	26.4	279.9	8.8	36.2	71.3	8.8	279.9
PCB 126	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 128	19.9	4.8	4.4	81.2	0.8	4.8	15.5	0.8	81.2
PCB 167	2.7	0.5	0.6	3.7	0.1	0.6	2.2	0.1	3.7
PCB 156	2.9	nd	0.3	3.3	nd	2.9	1.5	0.3	3.3
PCB 157	0.6	0.4	0.3	4.4	nd	0.5	1.2	0.3	4.4
PCB 169	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 187	14.2	5.0	4.4	24.7	4.1	5.0	9.8	4.1	24.7
PCB 180	55.4	11.0	11.2	133.0	8.2	11.2	44.3	8.2	133.0
PCB 170	14.3	3.0	3.3	47.9	1.8	3.3	11.2	1.8	47.9
PCB 189	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 195	1.6	0.3	0.7	4.8	0.5	0.7	1.1	0.3	4.8
PCB 206	1.0	nd	0.6	2.6	1.0	1.0	0.5	0.6	2.6
PCB 209	0.4	0.4	0.3	0.5	0.7	0.4	0.1	0.3	0.7
Sum PCB	674.3	305.4	132.1	1833.1	87.7	305.4	542.2	87.7	1833.1

2. PCB concentration (lipid weights) of Steller sea lion blubber samples (all locations)

Location ID Congeners	ng/g lipid weight Olutorsky Gulf (Russia) 1-Ageev Blubber	2-Ageev Blubber	3-Ageev Blubber	1-Testin Blubber	2-Testin Blubber	3-Testin Blubber	1-Udalov Blubber	2- Udalov Blubber	3-Udalov Blubber	1-Vozikov Blubber	Median	Interquartile Range	Range Min	Range Max
PCB 8	0.8	1.1	0.9	0.9	1.1	1.1	1.4	0.9	0.9	1.5	1.0	0.2	0.9	1.5
PCB 18	16.5	10.1	3.0	3.3	12.1	4.9	10.1	5.8	3.2	7.7	6.8	6.4	3.0	12.1
PCB 28	27.9	21.5	21.0	18.0	21.9	8.2	17.6	5.1	14.4	25.5	19.5	6.6	5.1	25.5
PCB 52	256.9	122.4	48.5	32.6	112.4	51.9	142.4	96.3	34.1	57.7	77.0	70.5	32.6	142.4
PCB 44	67.1	38.3	10.8	7.7	36.4	16.4	45.5	26.0	6.9	15.5	21.2	25.9	6.9	45.5
PCB 66	91,7	46.7	42.0	34.3	46.7	23.2	49.6	14.8	42.7	43.7	42.7	12.3	14.8	49.6
PCB 81	nd	nd	nd	nd	nd	nd	0.1	nd	nd	nd	0.1	0.0	0.1	0.1
PCB 77	1.7	2.6	nd	nd	0.9	0.5	2.0	1.3	nd	209.9	1.7	1.2	0.5	209.9
PCB 101	313.7	142.2	19.4	20.0	107.0	69.8	162.9	120.5	13.7	29.7	88.4	114.3	13.7	162.9
PCB 123	317.2	186.7	8.4	nd	156.1	116.2	118.4	137.7	178.9	nd	146.9	63.0	8.4	186.7
PCB 118	306.6	180.1	198.1	133.5	149.2	113.8	160.1	130.0	171.1	168.6	164.4	40.4	113.8	198.1
PCB 114	10.6	2.9	4.2	1.5	4.0	2.1	4.1	2.6	5.2	4.8	4.1	2.1	1.5	5.2
PCB 105	100.0	48.5	52.4	43.1	44.4	24.8	55.5	35.3	48.2	56.4	48.4	11.3	24.8	56.4
PCB 153	778.1	356.7	468.1	293.1	307.9	193.8	339.4	291.2	366.9	288.8	323.7	72.7	193.8	468.1
PCB 138	493.2	245.3	214.2	124.6	200.8	132.2	212.5	199.9	226.4	173.5	206.6	43.3	124.6	245.3
PCB 126	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 128	75.6	39.1	49.9	34.0	30.7	19.4	39.5	30.2	34.8	35.7	35.3	7.8	19.4	49.9
PCB 167	4.5	1.9	2.7	2.5	2.4	1.6	1.9	nd	3.5	2.6	2.5	0.8	1.6	3.5
PCB 156	5.5	1.3	3.6	2.2	3.2	1.4	1.8	nd	4.5	2.4	2.4	1.7	1.3	4.5
PCB 157	7.5	2.6	0.3	2.5	3.4	1.8	3.2	1.0	4.3	1.2	2.5	2.0	0.3	4.3
PCB 169	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 187	103.3	38.9	11.7	8.9	25.0	16.4	46.1	43.7	7.5	11.7	20.7	30.8	7.5	46.1
PCB 180	160.2	79.3	100.2	60.2	64.5	44.4	83.8	77.7	80.6	58.8	78.5	21.7	44.4	100.2
PCB 170	55.8	24.5	28.5	18.3	19.8	13.9	25.2	22.3	15.5	11.9	21.0	8.8	11.9	28.5
PCB 189	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 195	3.3	1.3	3.7	2.6	1.2	0.7	1.7	1.3	1.4	2.2	1.5	1.3	0.7	3.7
PCB 206	2.8	1.3	4.1	1.6	1.4	0.7	1.7	1.6	1.4	2.0	1.6	0.6	0.7	4.1
PCB 209	1.8	1.1	2.5	1.4	1.1	0.6	1.5	1.4	1.0	1.4	1.4	0.4	0.6	2.5
Sum PCB	3110.6	1596.1	1298.2	846.9	1353.7	859.7	1527.9	1246.3	1267.3	1213.2	1282.8	262.9	846.9	1596.1

Location ID Congeners	ng/g lipid weight St. Paul SSLSNP 2001-03 Blubber	SNPSLS9804 Blubber	SNPSLS9801 Blubber	SSLSNP 2001-05 Blubber	SNPSLS9803 Blubber	SNPSLS9902 Blubber	Median	Interquartile Range	Range Min	Range Max
PCB 8	4.6	1.9	2.3	1.1	4.3	3.5	2.9	2.1	1.1	4.6
PCB 18	7.5	4.8	5.1	2.2	7.1	8.4	6.1	2.5	2.2	8.4
PCB 28	21.6	15.2	28.3	9.0	61.6	41.7	24.9	21.5	9.0	61.6
PCB 52	50.4	77.3	76.3	16.9	132.0	60.3	68.3	24.1	16.9	132.0
PCB 44	11.3	13.2	7.0	2.5	21.8	11.9	11.6	4.8	2.5	21.8
PCB 66	120.9	77.4	118.9	139.7	175.1	97.4	119.9	32.2	77.4	175.1
PCB 81	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 77	0.9	nd	nd	nd	4.6	nd	2.7	1.9	0.9	4.6
PCB 101	29.5	64.4	42.2	19.1	61.3	32.1	37.2	26.4	19.1	64.4
PCB 123	5.7	nd	nd	nd	nd	115.5	60.6	54.9	5.7	115.5
PCB 118	609.4	257.2	384.3	613.5	537.5	392.1	464.8	205.1	257.2	613.5
PCB 114	14.1	6.9	10.1	20.6	0.0	8.7	9.4	5.7	0.0	20.6
PCB 105	176.4	77.2	129.6	201.8	179.5	154.7	165.6	42.8	77.2	201.8
PCB 153	1395.4	532.7	811.6	1270.2	972.6	1132.0	1052.3	383.8	532.7	1395.4
PCB 138	739.0	349.5	468.4	843.4	560.7	574.6	567.7	206.4	349.5	843.4
PCB 126	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 128	160.7	68.7	99.5	154.9	140.3	139.7	140.0	41.6	68.7	160.7
PCB 167	16.3	6.3	9.7	10.5	15.6	6.8	10.1	6.8	6.3	16.3
PCB 156	24.4	6.4	12.4	7.8	14.3	5.8	10.1	7.1	5.8	24.4
PCB 157	21.4	nd	nd	9.3	2.2	1.6	5.7	10.3	1.6	21.4
PCB 169	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 187	34.5	39.7	34.1	28.4	42.1	22.1	34.3	8.5	22.1	42.1
PCB 180	292.0	154.0	192.5	219.1	210.8	214.6	212.7	20.9	154.0	292.0
PCB 170	102.6	53.3	61.5	74.8	68.6	78.3	71.7	14.1	53.3	102.6
PCB 189	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 195	6.7	3.2	3.2	3.1	7.2	8.8	5.0	3.8	3.1	8.8
PCB 206	7.9	3.1	5.0	2.2	5.8	5.4	5.2	2.1	2.2	7.9
PCB 209	3.2	1.3	1.7	1.3	2.7	1.7	1.7	1.1	1.3	3.2
Sum PCB	3856.1	1813.7	2503.9	3651.4	3227.6	3117.6	3172.6	888.2	1813.7	3856.1

Location ID Congeners	ng/g lipid weight Tatitlek (males) Tatitlek # 4 Blubber	Tatitlek # 9 Blubber	Tatitlek # 3 Blubber	Tatitlek # 1 Blubber	Tatitlek #10 Blubber	Tatitlek #7 Blubber	Tatitlek #8 Blubber	Median	Interquartile Range	Range Min	 Max
PCB 8	2.6	3.9	0.5	nd	2.2	2.7	3.8	2.6	1.2	0.5	3.9
PCB 18	6.9	10.4	0.9	3.8	10.9	6.2	10.6	6.9	5.5	0.9	10.9
PCB 28	39.6	11.6	8.0	12.7	41.0	20.2	37.8	20.2	26.6	8.0	41.0
PCB 52	74.6	106.5	25.4	53.1	65.6	37.8	98.1	65.6	40.9	25.4	106.5
PCB 44	9.7	21.5	2.6	10.0	18.1	10.7	19.7	10.7	9.1	2.6	21.5
PCB 66	35.5	21.1	34.0	35.4	23.9	20.5	40.0	34.0	12.9	20.5	40.0
PCB 81	0.5	0.5	nd	nd	nd	nd	nd	0.5	0.0	0.5	0.5
PCB 77	nd	nd	nd	nd	0.9	nd	nd	0.9	0.0	0.9	0.9
PCB 101	63.2	136.5	31.4	50.6	42.4	22.8	55.9	50.6	22.6	22.8	136.5
PCB 123	nd	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 118	117.8	120.8	132.1	149.4	76.4	43.0	118.2	118.2	29.3	43.0	149.4
PCB 114	3.7	5.2	4.6	5.3	2.5	1.1	1.4	3.7	2.9	1.1	5.3
PCB 105	25.1	36.7	44.0	47.2	23.6	12.2	16.8	25.1	20.2	12.2	47.2
PCB 153	164.7	425.8	277.3	449.2	182.4	89.9	291.1	277.3	184.9	89.9	449.2
PCB 138	121.9	239.0	187.9	268.7	109.2	59.4	157.9	157.9	97.9	59.4	268.7
PCB 126	nd	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 128	16.1	36.8	27.7	37.4	17.0	9.3	25.7	25.7	15.7	9.3	37.4
PCB 167	2.6	1.4	4.5	4.2	nd	0.8	2.1	2.3	2.2	0.8	4.5
PCB 156	0.8	0.7	4.3	4.0	1.5	nd	nd	1.5	3.2	0.7	4.3
PCB 157	nd	0.8	2.4	4.0	1.2	0.2	3.6	1.8	2.4	0.2	4.0
PCB 169	nd	nd	nd	nd	nd	d	d	nd	nd	0.0	0.0
PCB 187	20.3	56.5	32.3	44.3	22.5	9.3	29.8	29.8	16.9	9.3	56.5
PCB 180	36.6	92.4	84.0	181.9	43.9	20.1	59.1	59.1	47.9	20.1	181.9
PCB 170	12.5	27.5	26.4	61.4	12.4	6.9	17.3	17.3	14.5	6.9	61.4
PCB 189	nd	nd	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 195	0.5	1.4	2.7	5.4	2.5	0.9	1.0	1.4	1.7	0.5	5.4
PCB 206	nd	2.2	3.9	3.2	nd	nd	2.1	2.7	1.2	2.1	3.9
PCB 209	nd	1.3	1.9	1.3	0.4	0.5	nd	1.3	0.8	0.4	1.9
Sum PCB	755.3	1360.6	938.6	1432.5	700.7	374.4	991.8	938.6	448.2	374.4	1432.5

Location ID Congeners	ng/g lipid weight Tatitlek (females) Tatitlek #5 Blubber	Tatitlek #6 Blubber	Tatitlek 3 Blubber	Tatitlek 1 Blubber	Tatitlek #2 Blubber	Median ng/g	Interquartile Range	Range Min	 Max
PCB 8	3.9	9.4	0.3	3.9	1.5	3.9	2.4	0.3	9.4
PCB 18	6.3	16.0	0.5	6.5	1.7	6.3	4.7	0.5	16.0
PCB 28	18.1	51.9	1.6	51.0	10.4	18.1	40.5	1.6	51.9
PCB 52	39.6	49.5	4.6	70.2	11.9	39.6	37.6	4.6	70.2
PCB 44	12.7	24.2	0.7	15.4	6.0	12.7	9.5	0.7	24.2
PCB 66	38.7	30.9	5.8	96.3	3.9	30.9	32.8	3.9	96.3
PCB 81	0.2	nd	nd	nd	nd	0.2	0.0	0.2	0.2
PCB 77	46.9	1.5	nd	nd	nd	24.2	22.7	1.5	46.9
PCB 101	41.2	51.1	6.8	49.5	10.5	41.2	38.9	6.8	51.1
PCB 123	nd	nd	nd	48.6	nd	48.6	0.0	48.6	48.6
PCB 118	118.1	59.6	29.7	279.1	6.3	59.6	88.4	6.3	279.1
PCB 114	2.7	1.8	0.6	7.4	0.1	1.8	2.1	0.1	7.4
PCB 105	37.8	14.6	8.3	113.5	1.3	14.6	29.4	1.3	113.5
PCB 153	260.5	109.7	56.6	837.0	18.8	109.7	203.8	18.8	837.0
PCB 138	131.9	62.5	38.2	354.3	10.4	62.5	93.7	10.4	354.3
PCB 126	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 128	26.9	8.2	6.4	102.8	0.9	8.2	20.6	0.9	102.8
PCB 167	3.7	0.9	0.9	4.7	0.1	0.9	2.8	0.1	4.7
PCB 156	3.9	nd	0.4	4.2	nd	3.9	1.9	0.4	4.2
PCB 157	0.9	0.7	0.4	5.6	nd	0.8	1.4	0.4	5.6
PCB 169	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 187	19.2	8.7	6.4	31.3	4.8	8.7	12.8	4.8	31.3
PCB 180	74.8	19.0	16.2	168.4	9.7	19.0	58.6	9.7	168.4
PCB 170	19.3	5.2	4.7	60.6	2.1	5.2	14.5	2.1	60.6
PCB 189	nd	nd	nd	nd	nd	nd	nd	0.0	0.0
PCB 195	2.2	0.6	1.1	6.1	0.6	1.1	1.6	0.6	6.1
PCB 206	1.3	nd	0.8	3.3	1.2	1.3	0.7	0.8	3.3
PCB 209	0.5	0.7	0.4	0.7	0.9	0.7	0.2	0.4	0.9
Sum PCB	911.2	526.6	191.4	2320.4	103.1	526.6	719.8	103.1	2320.4

3. PCB concentration (dry weights) of Steller sea lion liver samples (all locations)

ng/g dry weight

Location ID Congeners	St. Paul SSLSNP 2001-03 Liver	SSLSNP 2001-05 Liver	SNPSLS9801 Liver	SNPSLS9802 Liver	SNPSLS9804 Liver	Median	Interquartile Range	Range Min	Range Max
PCB 8	0.11	0.18	0.24	0.17	0.12	0.2	0.1	0.1	0.2
PCB 18	0.10	0.13	0.14	0.13	0.15	0.1	0.0	0.1	0.2
PCB 28	0.00	0.00	0.60	0.50	0.46	0.5	0.5	0.0	0.6
PCB 52	0.30	0.16	1.72	1.67	1.48	1.5	1.4	0.2	1.7
PCB 44	0.00	0.00	0.22	0.18	0.41	0.2	0.2	0.0	0.4
PCB 66	1.09	1.45	2.35	3.25	2.15	2.1	0.9	1.1	3.2
PCB 81	0.44	0.00	0.00	0.37	0.00	0.0	0.4	0.0	0.4
PCB 77	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 101	0.00	0.00	0.29	1.18	0.93	0.3	0.9	0.0	1.2
PCB 123	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 118	3.89	3.17	3.97	4.46	5.43	4.0	0.6	3.2	5.4
PCB 114	0.11	0.09	0.11	0.13	0.08	0.1	0.0	0.1	0.1
PCB 105	1.82	1.50	2.22	3.03	2.46	2.2	0.6	1.5	3.0
PCB 153	7.85	4.29	8.12	8.58	9.03	8.1	0.7	4.3	9.0
PCB 138	5.99	3.43	7.62	10.10	8.78	7.6	2.8	3.4	10.1
PCB 126	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 128	1.63	0.93	2.02	3.54	1.96	2.0	0.4	0.9	3.5
PCB 167	0.00	0.00	0.09	0.00	0.20	0.0	0.1	0.0	0.2
PCB 156	0.00	0.00	0.13	0.40	0.04	0.0	0.1	0.0	0.4
PCB 157	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 169	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 187	0.24	0.13	0.65	2.39	1.19	0.7	1.0	0.1	2.4
PCB 180	1.89	0.79	2.07	3.51	3.44	2.1	1.6	0.8	3.5
PCB 170	0.00	0.00	0.00	1.64	1.33	0.0	1.3	0.0	1.6
PCB 189	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 195	0.05	0.00	0.07	0.15	0.07	0.1	0.0	0.0	0.1
PCB 206	0.09	0.00	0.07	0.17	0.07	0.1	0.0	0.0	0.2
PCB 209	0.09	0.00	0.07	0.16	0.05	0.1	0.0	0.0	0.2
Sum PCB	25.7	16.2	32.8	45.7	39.9	32.8	14.2	16.2	45.7

Location ID Congeners	ng/g dry weight Tatitek (males) Tatitlek # 1 (7-4-02) Liver	Tatitlek #3 (4-25-03) Liver	Tatitlek #4 Liver	Tatitlek #7 Liver	Tatitlek #8 Liver	Tatitlek #10 Liver	Median	Interquartile Range	Range Min	Range Max
PCB 8	0.21	0.09	0.11	0.22	0.14	0.18	0.2	0.1	0.1	0.2
PCB 18	0.00	0.00	0.13	0.17	0.12	0.19	0.1	0.1	0.0	0.2
PCB 28	0.44	0.11	0.43	0.49	0.28	0.42	0.4	0.1	0.1	0.5
PCB 52	0.93	0.74	0.98	0.70	0.40	0.44	0.7	0.4	0.4	1.0
PCB 44	0.25	0.19	0.18	0.19	0.00	0.00	0.2	0.2	0.0	0.2
PCB 66	0.69	0.74	0.55	0.44	0.25	0.30	0.5	0.3	0.2	0.7
PCB 81	0.00	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 77	0.00	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 101	0.81	-0.05	0.50	0.43	0.07	0.21	0.3	0.4	0.0	0.8
PCB 123	0.00	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 118	1.94	1.71	1.12	1.09	0.28	0.51	1.1	0.9	0.3	1.9
PCB 114	0.00	0.04	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 105	0.93	0.73	0.44	0.32	0.08	0.17	0.4	0.5	0.1	0.9
PCB 153	5.63	2.90	2.53	2.03	0.65	0.97	2.3	1.6	0.7	5.6
PCB 138	4.68	3.06	1.63	1.24	0.39	0.79	1.4	1.8	0.4	4.7
PCB 126	0.23	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.2
PCB 128	0.89	0.64	0.28	0.17	0.06	0.12	0.2	0.4	0.1	0.9
PCB 167	0.00	0.00	0.03	0.05	0.00	0.00	0.0	0.0	0.0	0.1
PCB 156	0.12	0.00	0.05	0.00	0.00	0.00	0.0	0.0	0.0	0.1
PCB 157	0.00	0.00	0.03	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 169	0.00	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 187	1.47	0.83	0.41	0.22	0.08	0.18	0.3	0.5	0.1	1.5
PCB 180	3.44	1.21	0.55	0.58	0.15	0.30	0.6	0.7	0.2	3.4
PCB 170	1.04	0.00	0.00	0.00	0.05	0.00	0.0	0.0	0.0	1.0
PCB 189	0.00	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 195	0.00	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 206	0.05	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.1
PCB 209	0.04	0.04	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
Sum PCB	23.8	13.0	9.9	8.3	3.0	4.8	9.1	6.6	3.0	23.8

Location ID Congeners	Tatitek (females) Tatitlek #5 Liver	Tatitlek #6 Liver	Tatitlek #3 (2-2-01) Liver	Tatitlek #1 (28-10-00) Liver	Tatitlek #2 Liver	Median	Interquartile Range	Range Min	 Max
PCB 8	0.17	0.00	0.19	0.27	0.27	0.2	0.1	0.0	0.3
PCB 18	0.16	0.16	0.18	0.23	0.00	0.2	0.0	0.0	0.2
PCB 28	0.36	0.38	0.51	0.68	0.44	0.4	0.1	0.4	0.7
PCB 52	0.42	0.43	0.32	0.71	0.47	0.4	0.0	0.3	0.7
PCB 44	0.00	0.00	0.00	0.21	0.19	0.0	0.2	0.0	0.2
PCB 66	0.69	0.42	1.32	1.23	0.15	0.7	0.8	0.2	1.3
PCB 81	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 77	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 101	0.23	0.27	0.00	0.39	0.35	0.3	0.1	0.0	0.4
PCB 123	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 118	1.67	0.95	3.57	3.39	0.30	1.7	2.4	0.3	3.6
PCB 114	0.00	0.00	0.09	0.07	0.00	0.0	0.1	0.0	0.1
PCB 105	0.57	0.31	1.04	1.49	0.13	0.6	0.7	0.1	1.5
PCB 153	3.35	1.86	6.03	7.63	0.94	3.3	4.2	0.9	7.6
PCB 138	2.16	1.21	3.55	5.21	0.45	2.2	2.3	0.5	5.2
PCB 126	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 128	0.43	0.13	0.69	1.19	0.04	0.4	0.6	0.0	1.2
PCB 167	0.00	0.00	0.00	0.07	0.00	0.0	0.0	0.0	0.1
PCB 156	0.11	0.00	0.06	0.04	0.00	0.0	0.1	0.0	0.1
PCB 157	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 169	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 187	0.24	0.22	0.19	0.51	0.31	0.2	0.1	0.2	0.5
PCB 180	0.97	0.52	1.34	2.30	0.22	1.0	0.8	0.2	2.3
PCB 170	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 189	0.00	0.00	0.00	0.00	0.00	0.0	0.0	0.0	0.0
PCB 195	0.00	0.00	0.00	0.06	0.00	0.0	0.0	0.0	0.1
PCB 206	0.00	0.00	0.00	0.10	0.00	0.0	0.0	0.0	0.1
PCB 209	0.00	0.00	0.00	0.06	0.00	0.0	0.0	0.0	0.1
Sum PCB	11.5	6.9	19.1	25.8	4.3	11.5	12.2	4.3	25.8

4. PCB concentration (lipid weights) of Steller sea lion liver samples (all locations)

Location ID Congeners	ng/g lipid weight St. Paul SSLSNP 2001-03 Liver	SSLSNP 2001-05 Liver	SNPSLS9801 Liver	SNPSLS9802 Liver	SNPSLS9804 Liver	Median	Interquartile Range	Range Min	Range Max
PCB 8	1.4	2.2	4.3	3.7	2.0	2.2	1.7	1.4	4.3
PCB 18	1.2	1.5	2.5	2.8	2.6	2.5	1.0	1.2	2.8
PCB 28	0.0	0.0	10.8	11.0	7.8	7.8	10.8	0.0	11.0
PCB 52	3.7	1.9	31.0	36.8	25.0	25.0	27.3	1.9	36.8
PCB 44	0.0	0.0	4.0	3.9	6.9	3.9	4.0	0.0	6.9
PCB 66	13.2	17.7	42.3	71.7	36.3	36.3	24.6	13.2	71.7
PCB 81	5.3	0.0	0.0	8.2	0.0	0.0	5.3	0.0	8.2
PCB 77	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 101	0.0	0.0	5.3	26.0	15.7	5.3	15.7	0.0	26.0
PCB 123	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 118	47.3	38.6	71.6	98.3	91.8	71.6	44.5	38.6	98.3
PCB 114	1.3	1.1	2.1	2.9	1.4	1.4	0.8	1.1	2.9
PCB 105	22.1	18.3	40.0	66.7	41.6	40.0	19.5	18.3	66.7
PCB 153	95.3	52.3	146.3	189.3	152.6	146.3	57.3	52.3	189.3
PCB 138	72.7	41.7	137.4	222.7	148.4	137.4	73.7	41.7	222.7
PCB 126	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 128	19.7	11.3	36.4	78.0	33.1	33.1	16.7	11.3	78.0
PCB 167	0.0	0.0	1.6	0.0	3.3	0.0	1.6	0.0	3.3
PCB 156	0.0	0.0	2.4	8.9	0.7	0.7	2.4	0.0	8.9
PCB 157	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 169	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 187	2.9	1.5	11.8	52.8	20.2	11.8	17.2	1.5	52.8
PCB 180	22.9	9.6	37.4	77.4	58.1	37.4	35.2	9.6	77.4
PCB 170	0.0	0.0	0.0	36.2	22.4	0.0	22.4	0.0	36.2
PCB 189	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 195	0.6	0.0	1.2	3.2	1.1	1.1	0.6	0.0	3.2
PCB 206	1.2	0.0	1.2	3.8	1.2	1.2	0.1	0.0	3.8
PCB 209	1.0	0.0	1.3	3.5	0.9	1.0	0.4	0.0	3.5
Sum PCB	311.6	197.8	590.8	1007.9	673.2	590.8	361.5	197.8	1007.9

Location ID Congeners	ng/g lipid weight Tatitek (males) Tatitlek # 1 (7-4-02) Liver	Tatitlek #3 (4-25-03) Liver	Tatitlek #4 Liver	Tatitlek #7 Liver	Tatitlek #8 Liver	Tatitlek #10 Liver	Median	Interquartile Range	Range Min	Range Max
PCB 8	1.1	1.8	2.8	5.1	3.8	2.1	2.4	1.6	1.1	5.1
PCB 18	0.0	2.3	2.2	4.5	3.9	0.0	2.2	3.0	0.0	4.5
PCB 28	1.3	7.3	6.3	10.5	8.8	4.4	6.8	3.5	1.3	10.5
PCB 52	8.9	16.7	8.9	15.0	9.2	9.3	9.3	4.6	8.9	16.7
PCB 44	2.3	3.0	2.5	0.0	0.0	2.5	2.4	1.9	0.0	3.0
PCB 66	8.9	9.3	5.7	9.3	6.3	6.9	7.9	2.7	5.7	9.3
PCB 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 77	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 101	0.0	8.5	5.5	2.5	4.4	8.1	4.9	4.5	0.0	8.5
PCB 123	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 118	20.6	19.1	13.9	10.3	10.7	19.4	16.5	7.8	10.3	20.6
PCB 114	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4
PCB 105	8.8	7.5	4.0	3.1	3.6	9.3	5.8	4.8	3.1	9.3
PCB 153	34.9	43.0	25.9	24.2	20.4	56.3	30.4	16.4	20.4	56.3
PCB 138	36.7	27.9	15.9	14.7	16.7	46.8	22.3	18.4	14.7	46.8
PCB 126	0.0	0.0	0.0	0.0	0.0	2.3	0.0	0.0	0.0	2.3
PCB 128	7.7	4.8	2.2	2.3	2.5	8.9	3.6	4.6	2.2	8.9
PCB 167	0.0	0.5	0.7	0.0	0.0	0.0	0.0	0.4	0.0	0.7
PCB 156	0.0	0.8	0.0	0.0	0.0	1.2	0.0	0.6	0.0	1.2
PCB 157	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5
PCB 169	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 187	10.0	7.1	2.8	3.1	3.9	14.7	5.5	6.0	2.8	14.7
PCB 180	14.6	9.3	7.4	5.7	6.2	34.4	8.4	6.7	5.7	34.4
PCB 170	0.0	0.0	0.0	1.8	0.0	10.4	0.0	1.4	0.0	10.4
PCB 189	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 195	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 206	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.5
PCB 209	0.5	0.0	0.0	0.0	0.0	0.4	0.0	0.3	0.0	0.5
Sum PCB	156.5	169.4	106.7	112.2	100.3	238.0	134.4	58.1	100.3	238.0

Location ID Congeners	Tatitek (females) Tatitlek #5 Liver	Tatitlek #6 Liver	Tatitlek #3 (2-2-01) Liver	Tatitlek #1 (28-10-00) Liver	Tatitlek #2 Liver	Median	Interquartile Range	Range Min	Max
PCB 8	2.5	0.0	2.8	5.3	1.9	2.5	0.9	0.0	5.3
PCB 18	2.4	1.6	2.6	4.6	0.0	2.4	1.0	0.0	4.6
PCB 28	5.5	3.7	7.5	13.6	3.1	5.5	3.8	3.1	13.6
PCB 52	6.3	4.2	4.7	14.2	3.3	4.7	2.1	3.3	14.2
PCB 44	0.0	0.0	0.0	4.1	1.3	0.0	1.3	0.0	4.1
PCB 66	10.3	4.1	19.3	24.5	1.1	10.3	15.2	1.1	24.5
PCB 81	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 77	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 101	3.4	2.6	0.0	7.9	2.5	2.6	0.9	0.0	7.9
PCB 123	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 118	25.0	9.3	52.1	67.7	2.1	25.0	42.8	2.1	67.7
PCB 114	0.0	0.0	1.3	1.4	0.0	0.0	1.3	0.0	1.4
PCB 105	8.5	3.0	15.2	29.7	0.9	8.5	12.2	0.9	29.7
PCB 153	50.2	18.1	88.1	152.1	6.6	50.2	70.0	6.6	152.1
PCB 138	32.4	11.8	51.8	104.0	3.2	32.4	40.1	3.2	104.0
PCB 126	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 128	6.5	1.3	10.1	23.7	0.3	6.5	8.8	0.3	23.7
PCB 167	0.0	0.0	0.0	1.4	0.0	0.0	0.0	0.0	1.4
PCB 156	1.7	0.0	0.8	0.9	0.0	0.8	0.9	0.0	1.7
PCB 157	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 169	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 187	3.6	2.1	2.8	10.2	2.2	2.8	1.4	2.1	10.2
PCB 180	14.6	5.1	19.5	45.9	1.6	14.6	14.4	1.6	45.9
PCB 170	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 189	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PCB 195	0.0	0.0	0.0	1.2	0.0	0.0	0.0	0.0	1.2
PCB 206	0.0	0.0	0.0	2.0	0.0	0.0	0.0	0.0	2.0
PCB 209	0.0	0.0	0.0	1.1	0.0	0.0	0.0	0.0	1.1
Sum PCB	172.9	66.7	278.4	515.4	29.8	172.9	211.7	29.8	515.4

5. OCP concentration (dry weights) of Steller sea lion blubber samples (all locations)

Location	ng/g dry weight Olutorsky Gulf (Russia)													
Sample ID	1- Ageev	2- Ageev	3-Ageev	1-Testin	2-Testin	3-Testin	1-Udalov	2-Udalov	3-Udalov	1-Vozikov	Median	Interquartile	Range	
OCPs cong.	Blubber	Blubber	Blubber	Blubber	Blubber	Blubber	Blubber	Blubber	Blubber	Blubber		Range	Min	Max
a-BHC	35.9	40.5	21.4	17.2	44.9	27.5	26.6	24.2	31.8	24.4	27.0	10.6	17.2	44.9
B-BHC	370.0	123.7	118.9	118.6	205.2	82.6	176.2	113.7	107.9	131.9	121.3	50.2	82.6	370.0
r-BHC	19.5	14.2	9.5	5.3	21.3	11.2	11.2	10.5	15.1	12.1	11.6	4.2	5.3	21.3
8-BHC	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
HCB	1.8	1.2	2.0	1.3	1.5	2.4	1.2	1.8	3.4	1.6	1.7	0.6	1.2	3.4
Heptachlor	7.1	4.9	4.6	7.8	8.3	7.7	9.6	7.3	7.4	7.5	7.5	0.6	4.6	9.6
op'-DDE	12.6	9.8	0.6	1.1	9.0	5.4	7.4	6.9	1.0	2.1	6.2	7.3	0.6	12.6
pp'-DDE	1395.2	1190.3	418.5	305.0	965.5	642.6	833.4	1077.9	1064.3	430.7	899.4	590.8	305.0	1395.2
op'-DDD	36.6	26.9	5.2	3.4	30.0	18.3	21.7	19.2	10.5	10.4	18.7	15.2	3.4	36.6
pp'-DDD	380.5	161.0	57.6	70.3	183.0	88.2	169.3	156.8	108.7	78.1	132.8	86.6	57.6	380.5
op'-DDT	80.4	49.9	8.7	9.2	46.3	34.7	44.5	44.3	23.8	19.9	39.5	25.0	8.7	80.4
pp'-DDT	110.2	55.6	21.4	23.4	56.2	35.5	46.4	52.5	53.2	35.3	49.5	19.7	21.4	110.2

Location	ng/g dry weight Olutorsky Gulf (Russia)													
Sample ID	1- Ageev	2- Ageev	3-Ageev	1-Testin	2-Testin	3-Testin	1-Udalov	2-Udalov	3-Udalov	1-Vozikov	Median	Interquartile	Range	
Sum con.	Blubber	Blubber	Blubber	Blubber	Blubber	Blubber	Blubber	Blubber	Blubber	Blubber		Range	Min	Max
BHCs	425.3	178.4	149.9	141.1	271.4	121.3	214.0	148.4	154.8	168.4	161.6	56.4	121.3	425.3
HCB	1.8	1.2	2.0	1.3	1.5	2.4	1.2	1.8	3.4	1.6	1.7	0.6	1.2	3.4
Heptachlor	7.1	4.9	4.6	7.8	8.3	7.7	9.6	7.3	7.4	7.5	7.5	0.6	4.6	9.6
o,p DDTs	129.6	86.6	14.6	13.6	85.4	58.5	73.6	70.4	35.3	32.3	64.4	49.4	13.6	129.6
p,p DDTs	1885.9	1406.9	497.6	398.7	1204.7	766.4	1049.1	1287.2	1226.3	544.1	1126.9	672.3	398.7	1885.9

Location	ng/g dry weight St. Paul									
Sample ID	SSLSNP 2001-05	SNPSLS9803	SNPSLS9902	SSLSNP 2001-03	SNPSLS9804	SNPSLS9801	Median	Interquartile	Range	
OCPs conc.	Blubber	Blubber	Blubber	Blubber	Blubber	Blubber		Range	Min	Max
a-BHC	19.1	7.2	10.5	11.8	47.8	14.6	13.2	7.2	7.2	47.8
B-BHC	306.9	187.8	434.5	129.6	170.9	174.3	181.1	105.4	129.6	434.5
r-BHC	6.3	3.5	2.2	4.9	11.3	0.0	4.2	3.5	0.0	11.3
8-BHC	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
HCB	2.9	1.1	1.4	2.4	4.2	1.1	1.9	1.6	1.1	4.2
Heptachlor	11.3	3.6	19.2	10.8	14.0	20.0	12.6	7.0	3.6	20.0
op'-DDE	0.7	1.6	0.5	0.5	6.9	0.8	0.7	0.8	0.5	6.9
pp'-DDE	2790.2	688.0	623.5	2072.4	1731.1	1818.3	1774.7	1060.0	623.5	2790.2
op'-DDD	2.5	4.4	2.6	2.2	18.4	2.6	2.6	1.5	2.2	18.4
pp'-DDD	98.2	140.0	127.5	116.5	338.0	193.1	133.8	60.5	98.2	338.0
op'-DDT	10.2	37.3	5.6	7.0	79.0	17.8	14.0	24.7	5.6	79.0
pp'-DDT	385.5	119.8	36.6	43.9	248.6	91.0	105.4	160.8	36.6	385.5

Location	ng/g dry weight Olutorsky Gulf (Russia)									
Sample ID	SSLSNP 2001-05	SNPSLS9803	SNPSLS9902	SSLSNP 2001-03	SNPSLS9804	SNPSLS9801	Median	Interquartile	Range	
Sum con.	Blubber	Blubber	Blubber	Blubber	Blubber	Blubber		Range	Min	Max
BHCs	332.4	198.5	447.1	146.3	230.0	189.0	214.2	115.4	146.3	447.1
HCB	2.9	1.1	1.4	2.4	4.2	1.1	1.9	1.6	1.1	4.2
Heptachlor	11.3	3.6	19.2	10.8	14.0	20.0	12.6	7.0	3.6	20.0
o,p DDTs	13.4	43.3	8.7	9.6	104.4	21.2	17.3	27.2	8.7	104.4
p,p DDTs	3273.8	947.9	787.7	2232.8	2317.8	2102.5	2167.6	1060.0	787.7	3273.8

Location Sample ID OCPs cong.	ng/g dry weight Tatitlek (males) Tatitlek #1 Blubber	Tatitlek #3 Blubber	Tatitlek #9 Blubber	Tatitlek #4 Blubber	Tatitlek #10 Blubber	Tatitlek #7 Blubber	Tatitlek #8 Blubber	Median	Interquartile Range	Range Min	Range Max
a-BHC	45.0	14.1	5.4	38.4	28.1	17.1	28.1	28.1	17.7	5.4	45.0
B-BHC	120.1	102.1	70.6	71.4	53.4	33.2	57.5	70.6	31.3	33.2	120.1
r-BHC	12.5	2.4	4.3	8.6	2.5	5.5	9.5	5.5	5.7	2.4	12.5
8-BHC	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
HCB	2.4	3.3	1.7	1.0	0.9	0.7	0.7	1.0	1.2	0.7	3.3
Heptachlor	4.5	2.3	32.9	19.5	0.0	13.3	11.3	11.3	13.0	0.0	32.9
op'-DDE	4.1	0.8	6.3	2.5	1.1	0.4	1.1	1.1	2.4	0.4	6.3
pp'-DDE	1357.5	1432.8	1262.9	574.6	255.2	210.7	219.7	574.6	1072.7	210.7	1432.8
op'-DDD	17.5	2.9	10.2	15.7	6.1	3.5	6.7	6.7	8.2	2.9	17.5
pp'-DDD	127.8	76.1	122.9	94.3	54.3	42.9	48.7	76.1	57.1	42.9	127.8
op'-DDT	43.1	4.7	25.2	31.6	10.1	5.5	12.4	12.4	20.6	4.7	43.1
pp'-DDT	92.4	84.3	28.5	52.1	20.6	17.4	20.2	28.5	47.9	17.4	92.4

Location Sample ID Sum con.	ng/g dry weight Tatitlek (males) Tatitlek #1 Blubber	Tatitlek #3 Blubber	Tatitlek #9 Blubber	Tatitlek #4 Blubber	Tatitlek #10 Blubber	Tatitlek #7 Blubber	Tatitlek #8 Blubber	Median	Interquartile Range	Range Min	Range Max
BHCs	177.5	118.6	80.3	118.4	83.9	55.7	95.1	95.1	36.4	55.7	177.5
HCB	2.4	3.3	1.7	1.0	0.9	0.7	0.7	1.0	1.2	0.7	3.3
Heptachlor	4.5	2.3	32.9	19.5	0.0	13.3	11.3	11.3	13.0	0.0	32.9
o,p DDTs	64.7	8.5	41.8	49.8	17.3	9.4	20.1	20.1	32.4	8.5	64.7
p,p DDTs	1577.6	1593.2	1414.2	721.1	330.1	271.0	288.6	721.1	1186.6	271.0	1593.2

Location Sample ID OCPs cong.	Tatitlek(females) Tatitlek #2 Blubber	Tatitlek 1 Blubber	Tatitlek #5 Blubber	Tatitlek #6 Blubber	Tatitlek 3 Blubber	Median	Interquartile Range	Range Min	 Max
a-BHC	7.6	27.1	6.8	5.7	9.7	7.6	2.9	5.7	27.1
B-BHC	6.7	452.4	32.7	20.5	76.2	32.7	55.7	6.7	452.4
r-BHC	2.1	10.7	2.0	0.8	2.2	2.1	0.2	0.8	10.7
8-BHC	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
HCB	1.5	1.6	2.0	1.6	3.2	1.6	0.4	1.5	3.2
Heptachlor	6.2	10.7	12.3	16.2	3.2	10.7	6.0	3.2	16.2
op'-DDE	0.3	2.4	0.7	0.3	0.7	0.7	0.5	0.3	2.4
pp'-DDE	31.4	453.7	278.1	156.5	960.3	278.1	297.2	31.4	960.3
op'-DDD	1.1	2.6	1.7	0.8	2.6	1.7	1.5	0.8	2.6
pp'-DDD	8.9	275.7	35.3	16.0	68.9	35.3	52.9	8.9	275.7
op'-DDT	1.7	17.6	4.1	0.5	3.2	3.2	2.4	0.5	17.6
pp'-DDT	3.1	66.9	41.5	9.9	69.0	41.5	57.1	3.1	69.0

Location Sample ID Sum con.	Tatitlek(females) Tatitlek #2 Blubber	Tatitlek 1 Blubber	Tatitlek #5 Blubber	Tatitlek #6 Blubber	Tatitlek 3 Blubber	Median	Interquartile Range	Range Min	 Max
BHCs	16.3	490.2	41.5	26.9	88.0	41.5	61.1	16.3	490.2
HCB	1.5	1.6	2.0	1.6	3.2	1.6	0.4	1.5	3.2
Heptachlor	6.2	10.7	12.3	16.2	3.2	10.7	6.0	3.2	16.2
o,p DDTs	3.0	22.6	6.6	1.6	6.5	6.5	3.5	1.6	22.6
p,p DDTs	43.4	796.3	354.9	182.4	1098.3	354.9	614.0	43.4	1098.3

6. OCP concentration (lipid weights) of Steller sea lion blubber samples (all locations)

Location Sample ID OCPs conc.	ng/g lipid weight Olutorsky Gulf (Russia) 1- Ageev Blubber	2- Ageev Blubber	3-Ageev Blubber	1-Testin Blubber	2-Testin Blubber	3-Testin Blubber	1-Udalov Blubber	2-Udalov Blubber	3-Udalov Blubber	1-Vozikov Blubber	Median	Interquartile Range	Range Min	 Max
a-BHC	42.4	51.9	32.1	20.9	50.3	33.2	34.4	30.3	41.0	37.7	36.0	9.7	20.9	51.9
B-BHC	438.1	158.6	178.4	144.2	229.9	99.6	227.7	142.3	138.9	203.7	168.5	79.0	99.6	438.1
r-BHC	23.0	18.1	14.3	6.5	23.9	13.5	14.5	13.1	19.5	18.6	16.3	5.5	6.5	23.9
8-BHC	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
HCB	2.2	1.5	3.0	1.6	1.7	2.9	1.6	2.3	4.4	2.5	2.2	1.2	1.5	4.4
Heptachlor	8.4	6.3	6.9	9.5	9.3	9.2	12.5	9.1	9.6	11.6	9.3	1.0	6.3	12.5
op'-DDE	14.9	12.6	0.9	1.3	10.1	6.6	9.6	8.6	1.3	3.2	7.6	8.2	0.9	14.9
pp'-DDE	1652.1	1525.5	627.8	370.8	1082.1	775.2	1077.0	1349.0	1370.2	665.0	1079.6	672.3	370.8	1652.1
op'-DDD	43.3	34.4	7.8	4.1	33.7	22.1	28.0	24.0	13.5	16.0	23.1	18.1	4.1	43.3
pp'-DDD	450.5	206.4	86.4	85.5	205.1	106.4	218.8	196.3	140.0	120.7	168.1	96.1	85.5	450.5
op'-DDT	95.2	63.9	13.1	11.2	51.9	41.9	57.5	55.4	30.6	30.7	46.9	26.3	11.2	95.2
pp'-DDT	130.5	71.3	32.2	28.4	63.0	42.9	60.0	65.7	68.6	54.5	61.5	22.0	28.4	130.5

Location Sample ID OCPs conc.	ng/g lipid weight Olutorsky Gulf (Russia) 1- Ageev Blubber	2- Ageev Blubber	3-Ageev Blubber	1-Testin Blubber	2-Testin Blubber	3-Testin Blubber	1-Udalov Blubber	2-Udalov Blubber	3-Udalov Blubber	1-Vozikov Blubber	Median	Interquartile Range	Range Min	 Max
BHCs	503.6	228.6	224.8	171.6	304.1	146.3	276.6	185.7	199.3	260.0	226.7	83.4	146.3	503.6
HCB	2.2	1.5	3.0	1.6	1.7	2.9	1.6	2.3	4.4	2.5	2.2	1.2	1.5	4.4
Heptachlor	8.4	6.3	6.9	9.5	9.3	9.2	12.5	9.1	9.6	11.6	9.3	1.0	6.3	12.5
o,p DDTs	153.4	111.0	21.8	16.6	95.7	70.5	95.1	88.1	45.4	49.9	79.3	49.0	16.6	153.4
p,p DDTs	2386.5	1914.1	768.2	501.3	1445.9	995.0	1451.0	1699.0	1624.2	890.2	1448.4	763.9	501.3	2386.5

Location Sample ID OCPs conc.	ng/g lipid weight St. Paul SSLSNP 2001-05 Blubber	SNPSLS9803 Blubber	SNPSLS9902 Blubber	SSLSNP 2001-03 Blubber	SNPSLS9804 Blubber	SNPSLS9801 Blubber	Median	Interquartile Range	Range Min	 Max
a-BHC	36.8	20.6	14.3	22.6	57.6	21.6	22.1	12.4	14.3	57.6
B-BHC	590.2	536.6	595.2	249.3	205.9	256.8	396.7	325.6	205.9	595.2
r-BHC	12.2	9.9	3.0	9.5	13.6	0.0	9.7	7.0	0.0	13.6
8-BHC	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
HCB	5.6	3.2	1.9	4.5	5.1	1.7	3.8	2.7	1.7	5.6
Heptachlor	21.7	10.2	26.3	20.8	16.8	29.4	21.2	7.3	10.2	29.4
op'-DDE	1.3	4.5	0.7	0.9	8.4	1.1	1.2	2.7	0.7	8.4
pp'-DDE	5365.7	1965.9	854.2	3985.3	2085.7	2678.8	2382.2	1662.9	854.2	5365.7
op'-DDD	4.7	12.5	3.6	4.2	22.2	3.8	4.4	6.7	3.6	22.2
pp'-DDD	188.8	400.1	174.6	224.1	407.2	284.5	254.3	173.5	174.6	407.2
op'-DDT	19.7	106.7	7.6	13.4	95.2	26.3	23.0	63.0	7.6	106.7
pp'-DDT	741.3	342.4	50.2	84.4	299.6	134.1	216.8	234.9	50.2	741.3

Location Sample ID OCPs conc.	ng/g lipid weight St. Paul SSLSNP 2001-05 Blubber	SNPSLS9803 Blubber	SNPSLS9902 Blubber	SSLSNP 2001-03 Blubber	SNPSLS9804 Blubber	SNPSLS9801 Blubber	Median	Interquartile Range	Range Min	 Max
BHCs	639.1	567.1	612.5	281.4	277.1	278.4	424.2	322.0	277.1	639.1
HCB	5.6	3.2	1.9	4.5	5.1	1.7	3.8	2.7	1.7	5.6
Heptachlor	21.7	10.2	26.3	20.8	16.8	29.4	21.2	7.3	10.2	29.4
o,p DDTs	25.8	123.7	12.0	18.5	125.8	31.2	28.5	80.2	12.0	125.8
p,p DDTs	6321.6	2832.0	1091.0	4312.4	2918.3	3128.5	3023.4	1162.8	1091.0	6321.6

Location Sample ID OCPs cong.	ng/g lipid weight Tatitlek (males) Tatitlek #1 Blubber	Tatitlek #3 Blubber	Tatitlek #9 Blubber	Tatitlek #4 Blubber	Tatitlek #10 Blubber	Tatitlek #7 Blubber	Tatitlek #8 Blubber	Median	Interquartile Range	Range Min	 Max
a-BHC	55.5	17.3	5.1	47.5	53.9	24.8	69.6	47.5	33.7	5.1	69.6
ß-BHC	148.3	125.0	66.6	88.1	102.6	48.1	142.3	102.6	56.3	48.1	148.3
r-BHC	15.4	2.9	4.0	10.7	4.8	7.9	23.4	7.9	8.6	2.9	23.4
8-BHC	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
HCB	2.9	4.0	1.6	1.3	1.7	1.0	1.7	1.7	0.9	1.0	4.0
Heptachlor	5.5	2.8	31.1	24.1	0.0	19.2	27.9	19.2	21.8	0.0	31.1
op'-DDE	5.0	1.0	6.0	3.1	2.1	0.6	2.6	2.6	2.5	0.6	6.0
pp'-DDE	1675.9	1753.9	1191.4	709.4	490.8	305.3	543.5	709.4	916.4	305.3	1753.9
op'-DDD	21.7	3.6	9.6	19.3	11.7	5.0	16.5	11.7	10.6	3.6	21.7
pp'-DDD	157.7	93.1	115.9	116.5	104.4	62.2	120.4	115.9	19.7	62.2	157.7
op'-DDT	53.2	5.8	23.8	39.0	19.4	8.0	30.7	23.8	21.1	5.8	53.2
pp'-DDT	114.1	103.2	26.9	64.3	39.5	25.3	49.9	49.9	50.6	25.3	114.1

Location Sample ID OCPs cong.	ng/g lipid weight Tatitlek (males) Tatitlek #1 Blubber	Tatitlek #3 Blubber	Tatitlek #9 Blubber	Tatitlek #4 Blubber	Tatitlek #10 Blubber	Tatitlek #7 Blubber	Tatitlek #8 Blubber	Median	Interquartile Range	Range Min	 Max
BHCs	219.2	145.2	75.8	146.2	161.3	80.8	235.3	146.2	77.3	75.8	235.3
HCB	2.9	4.0	1.6	1.3	1.7	1.0	1.7	1.7	0.9	1.0	4.0
Heptachlor	5.5	2.8	31.1	24.1	0.0	19.2	27.9	19.2	21.8	0.0	31.1
o,p DDTs	79.9	10.3	39.4	61.4	33.2	13.6	49.8	39.4	32.2	10.3	79.9
p,p DDTs	2027.6	1960.6	1373.5	951.7	668.0	406.4	763.7	951.7	951.2	406.4	2027.6

Location Sample ID OCPs cong.	Tatitlek(females) Tatitlek #2 Blubber	Tatitlek 1 Blubber	Tatitlek #5 Blubber	Tatitlek #6 Blubber	Tatitlek 3 Blubber	Median	Interquartile Range	Range Min	 Max
a-BHC	8.9	34.3	9.2	9.8	14.0	9.8	4.9	8.9	34.3
B-BHC	7.8	572.6	44.2	35.3	110.3	44.2	75.0	7.8	572.6
r-BHC	2.4	13.6	2.7	1.3	3.1	2.7	0.7	1.3	13.6
8-BHC	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
HCB	1.8	2.1	2.7	2.8	4.7	2.7	0.7	1.8	4.7
Heptachlor	7.3	13.6	16.6	27.9	4.6	13.6	9.2	4.6	27.9
op'-DDE	0.3	3.0	1.0	0.4	1.0	1.0	0.6	0.3	3.0
pp'-DDE	36.8	574.4	375.8	269.8	1391.0	375.8	304.5	36.8	1391.0
op'-DDD	1.2	3.3	2.3	1.4	3.7	2.3	1.9	1.2	3.7
pp'-DDD	10.4	349.0	47.7	27.6	99.8	47.7	72.2	10.4	349.0
op'-DDT	2.0	22.3	5.6	0.9	4.6	4.6	3.6	0.9	22.3
pp'-DDT	3.7	84.7	56.1	17.0	100.0	56.1	67.7	3.7	100.0

Location Sample ID Sum con.	Tatitlek(females) Tatitlek #2 Blubber	Tatitlek 1 Blubber	Tatitlek #5 Blubber	Tatitlek #6 Blubber	Tatitlek 3 Blubber	Median	Interquartile Range	Range Min	 Max
BHCs	19.14	620.51	56.11	46.42	127.49	56.1	81.1	19.1	620.5
HCB	1.81	2.06	2.68	2.77	4.66	2.7	0.7	1.8	4.7
Heptachlor	7.31	13.56	16.56	27.88	4.56	13.6	9.2	4.6	27.9
o,p DDTs	3.56	28.55	8.88	2.69	9.36	8.9	5.8	2.7	28.5
p,p DDTs	54.45	1036.58	488.51	317.10	1600.15	488.5	719.5	54.4	1600.1

7. Lipid weights (%) of all Steller sea lion tissues (blubber and liver) from all locations

Location	Gender	Sample ID	Tissue	Lipid content %
Olutorsky Gulf (R)	male	1 Ageev	blubber	84.5
		2 Ageev	blubber	78.0
		3 Ageev	blubber	66.7
		1 Testin	blubber	82.3
		2 Testin	blubber	89.2
		3 Testin	blubber	82.9
		1 Udalov	blubber	77.4
		2 Udalov	blubber	79.9
		3 Udalov	blubber	77.7
		1 Vozikov	blubber	64.8
Tatitlek (PWS)	male	Tatitlek #8	blubber	40.4
			liver	3.0
		Tatitlek #10	blubber	52.0
			liver	5.0
		Tatitlek #3	blubber	81.7
			liver	8.0
		Tatitlek #7	blubber	69.0
			liver	8.0
		Tatitlek #4	blubber	81.0
			liver	6.0
		Tatitlek # 1	blubber	81.0
			liver	10.0
		Tatitlek #9	blubber	106.0
St. Paul (BS)	male	SNPSLS9801	blubber	67.9
			liver	6.0
		SSLSNP 2001-05	blubber	82.1
			liver	8.0
		SSLSNP 2001-03	blubber	52.0
			liver	8.0
		SNPSLS9803	blubber	35.0
		SNPSLS9804	blubber	83.0
			liver	6.0
		SNPSLS9902	blubber	73.0
		SNPSLS9802	liver	5.0
Tatitlek (PWS)	female	Tatitlek 1	blubber	79.0
			liver	5.0
		Tatitlek #6	blubber	58.0
			liver	10.0
		Tatitlek 3	blubber	69.0
			liver	7.0
		Tatitlek #5	blubber	74.0
			liver	7.0
		Tatitlek #2	blubber	85.3
			liver	14.0

PWS = Prince William Sound ; BS = Bering Sea ; R = Russia